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DEVELOPMENT OF METHODS TO DEFINE WATER QUALITY EFFECTS OF URBAN RUNOFF

by

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16. ABSTRACT

This project considers short-term water quality impacts that occur during or shortly after a storm event. Examples of the short term impacts are dissolved oxygen depressions due to rapid oxidation of contaminants or the death of fish as a result of short term increases in the concentration of a toxic in the receiving water. The phenomena which characterize these impacts are related to event characteristics such as the volume and duration of the runoff, the concentration of a contaminant in the runoff, and the dilution available in the receiving water during the runoff event. This dilution can be characterized, on the scale of the total river width, by the joint occurrence of storm discharges from urban areas and the stream flow in the receiving waters. The project addresses methods for defining the effects of time variable concentrations on organism mortality and includes considerations of carryover effects between storms as a result of varying instream contaminant concentrations during dry weather. The methods presented for evaluation of effects from exposure to time variable concentrations can be considered for use in defining wet weather water quality criteria.

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FOREWORD

The Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of that environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is that necessary first step in problem solution and it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems for the prevention, treatment, and management of wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, for the preservation and treatment of public drinking water supplies, and for minimizing the adverse economic, social, health, and aesthetic effects of pollution. This publication is one of the products of that research, a most vital communications link between the researcher and the user community.

This report summarizes the results of a study effort investigating methods which can be employed to develop wet weather water quality criteria which could form part of the basis for wet weather standards. The wet weather criteria could ultimately be employed to develop measures of benefits to be obtained from treatment of combined sewer overflows and urban runoff.

Francis T. Mayo, Director
Municipal Environmental
Research Laboratory

ABSTRACT

The projected costs for treating combined sewer overflows and urban runoff nationwide are extremely large, and therefore necessitate that methods be available to quantitatively evaluate the receiving water impacts associated with these discharges. This report summarizes the results of a study effort investigating methods which can be employed to develop wet weather water quality criteria which could form part of the basis for wet weather standards. The wet weather criteria could ultimately be employed to develop measures of benefits to be obtained from treatment of combined sewer overflows and urban runoff.

This project considers short-term water quality impacts that occur during or shortly after a storm event. Examples of the short term impacts are dissolved oxygen depressions due to rapid oxidation of contaminants or the death of fish as a result of short term increases in the concentration of a toxic in the receiving water. The phenomena which characterize these impacts are related to event characteristics such as the volume and duration of the runoff, the concentration of a contaminant in the runoff, and the dilution available in the receiving water during the runoff event. This dilution can be characterized, on the scale of the total river width, by the joint occurrence of storm discharges from urban areas and the stream flow in the receiving waters. The project addresses methods for defining the effects of time variable concentrations on organism mortality and includes considerations of carryover effects between storms as a result of varying instream contaminant concentrations during dry weather. The methods presented for evaluation of effect from exposure to time variable concentrations can be considered for use in defining wet weather water quality criteria.

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SECTION I

INTRODUCTION

Existing and possible future nationwide programs to control discharges from combined sewers and the runoff from separately sewered urban areas, can require capital costs that range from tens to hundreds of billions of dollars with associated operating costs of hundreds of millions of dollars annually. It is therefore necessary that methods be available to quantitatively evaluate the receiving water impacts associated with these discharges so that an estimate of benefits can be obtained and compared to the very substantial costs involved. One method that has been employed to indirectly assess receiving water benefits utilizes local water quality standards which consider economic and social impacts together with water quality criteria defined by the beneficial water use to be protected. This report presents the results of a project that has investigated methods which can be employed to develop wet weather water quality criteria which could form part of the basis for wet weather standards. The wet weather criteria could ultimately be employed to develop measures of benefits to be obtained from treatment of CSO and urban runoff.

There are two types of water quality problems which are normally associated with the discharges from combined sewers and urban runoff. The first water quality impact, which was studied, is characterized by rapid short term changes in water quality during and shortly after a storm event. The phenomena which characterize these impacts are related to event characteristics such as the volume of runoff, the duration of the runoff, the concentration of a contaminant in the runoff, and the dilution available in the receiving water during the runoff event. Examples of the short term impacts are dissolved oxygen depressions due to rapid oxidation of contaminants or the death of fish as a result of short term increases in the concentration of a toxicant in the receiving water.

The magnitude of water quality impacts from urban stormwater discharges has been defined in part by the dilution available in the receiving water. This dilution has been characterized, on the scale of the total river width, by the joint occurrence of storm discharges from urban areas and the stream flow in the receiving waters. A major area of investigation in the project addressed methods for defining the effects of time variable concentrations on organism mortality, and includes considerations of carryover effects between storms as a result of varying instream contaminant concentrations during dry weather.

In contrast to the short term impacts, combined sewer overflows and urban runoff can contribute to longer term water quality degradation. The long term impacts are caused by the contaminants which are associated with the suspended solids in the discharges or contaminants which become associated with solids in the receiving water. In this context long term impacts may also be associated with nutrients which are both dissolved and particulate. The long term impacts can be characterized by mass loading to receiving waters accumulated over periods of time extending from seasonally to annually. Examples of long term impacts are bottom oxygen demand of accumulated sediments or the biological accumulation of toxics as a result of leaching from sediments or uptake by benthic organisms. In the former example, for dissolved oxygen, the critical water quality impact is usually associated with the normal low flow-high temperature summer periods. In both of the examples cited above and for all other long term effects one basic measure of the effectiveness of control actions can be obtained from comparisons of mass loadings from the various sources. Conventional analysis techniques such as steady state modeling can then be employed to develop an estimate of water quality improvement and benefits. This latter situation has been addressed in the first year report. Water quality problems, associated with bottom scour of sediment, have not been considered in this project.

SECTION 2

CONCLUSIONS

1. A method has been developed which uses existing data from classical bioassay tests to calculate the effects of exposure to time variable concentrations of toxicants on aquatic organisms such as fish. The procedure calculates the "equivalent dosage" and can define the "equivalent mortality dosage". This procedure can be used to define water quality criteria for time variable event related phenomena such as those associated with urban discharges.

2. The procedure for calculation of "equivalent dosage" has been tested employing several sets of data collected independently.

3. A method has also been developed to define the effects on fish of low dissolved oxygen levels which occur on the short time scales associated with storm water discharges.

4. A calculation procedure has been presented which combines statistics on rainfall/runoff, stream flow, concentrations in streams, and concentrations in CSO and/or urban runoff to calculate the statistical distribution of instream concentrations of contaminants.

5. The methods for calculation of wet weather criteria and the statistics of instream concentrations can be combined to define locations where water quality problems are to be anticipated as a result of storm related discharges from urban areas.

6. Judgements on the extent of water quality problems which are associated with CSO and urban runoff discharges are dependent to a large extent on the water quality criteria which are used. If generalized water quality criteria, usually employed for continuous discharges, are considered CSO and urban runoff can produce water quality problems under most circumstances. If criteria are developed which consider the short duration of storm events and the relatively large interval between storms the circumstances where there are water quality problems associated with CSO and urban runoff appear to be significantly reduced.

SECTION 3

RECOMMENDATIONS

1. Additional testing with collection of appropriate data are suggested for:

   A. The methods defining the effects on aquatic organisms of time variable exposure to concentrations of toxics and low dissolved oxygen levels.

   B. The calculation procedure for defining the statistics of instream concentrations of contaminants.

2. The method for calculating "equivalent dosage" considering exposures to time variable concentrations of toxics should be employed to analyze a larger cross section of bioassay data.

SECTION 4

POLLUTANT LOADS

Data from the University of Florida(1) data base on the event mean concentrations observed during discharges from combined sewer and separate sewer systems were analyzed. Figures 1 to 4 present typical results of a log-normal probability analysis of the aggregated data from all sites for event mean concentrations for each type of sewer system. The event mean concentration for the sites available can usually be adequately represented by a log-normal probability distribution. The individual log normal probability plots for other contaminants are presented in the Appendix.

Since data aggregated from all sites can usually be represented by one log-normal distribution, this project considered an analysis which employs a single log-normal distribution of the event mean concentrations for individual contaminants. The median and variance estimated from the log-normal plots are summarized in Tables 1 and 2 for the constituents analyzed. These statistical parameters were employed in subsequent analysis of probable receiving water impacts from CSO and urban runoff.

The full implications associated with the ability to represent event mean concentration data from many sites with a single log-normal probability distribution is unclear at this time. If the event mean concentration data from all sites are from the same statistical population then use of a single distribution to represent this population provides a true estimate of the load and concentrations from CSO and urban runoff. If the data on event mean concentrations are from several statistically separate populations then use of a single distribution will tend to overestimate the frequency distribution of loads and concentrations associated with some populations and underestimate the frequency distributions of loads and concentrations for other populations. The existing data base on event mean concentrations is small considering the inherent variability in the data. However some preliminary analysis suggests that more than one population may be present in the event mean concentration data. In this case the use of a single distribution to represent all sites can be considered as an overall averaging of a total data base.

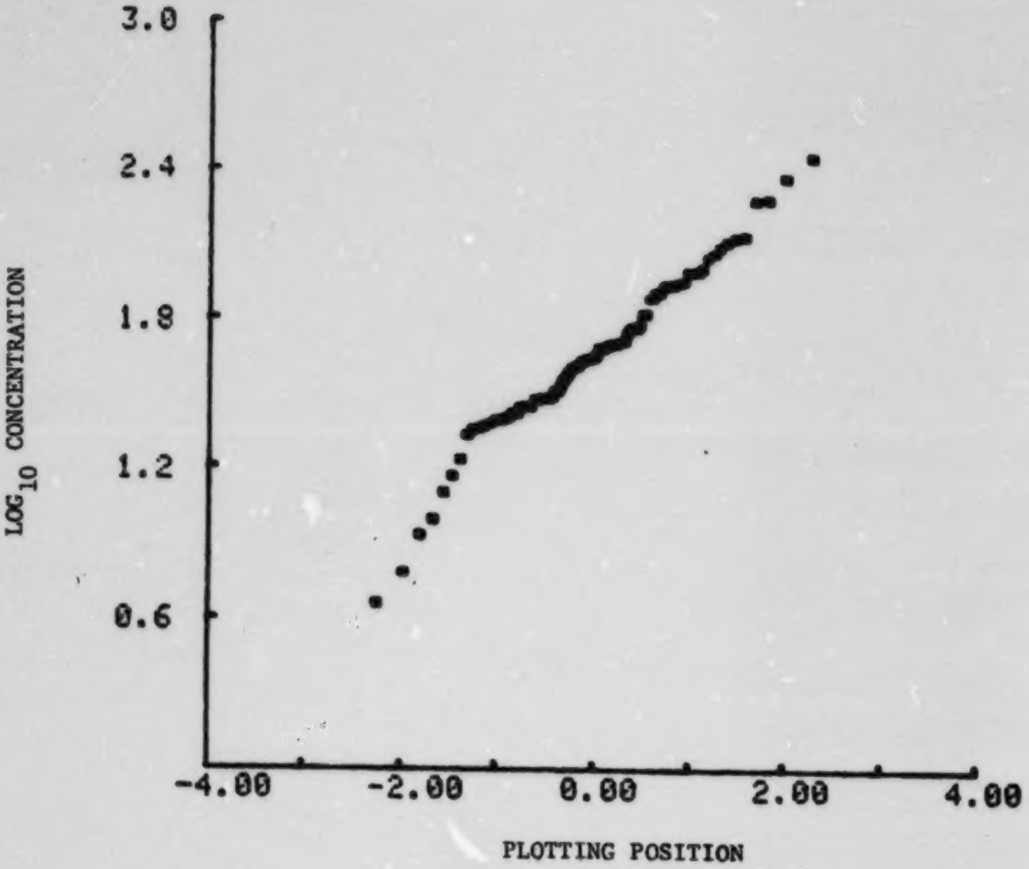

Figure 1. BOD Concentration Data from Combined Sewer Systems

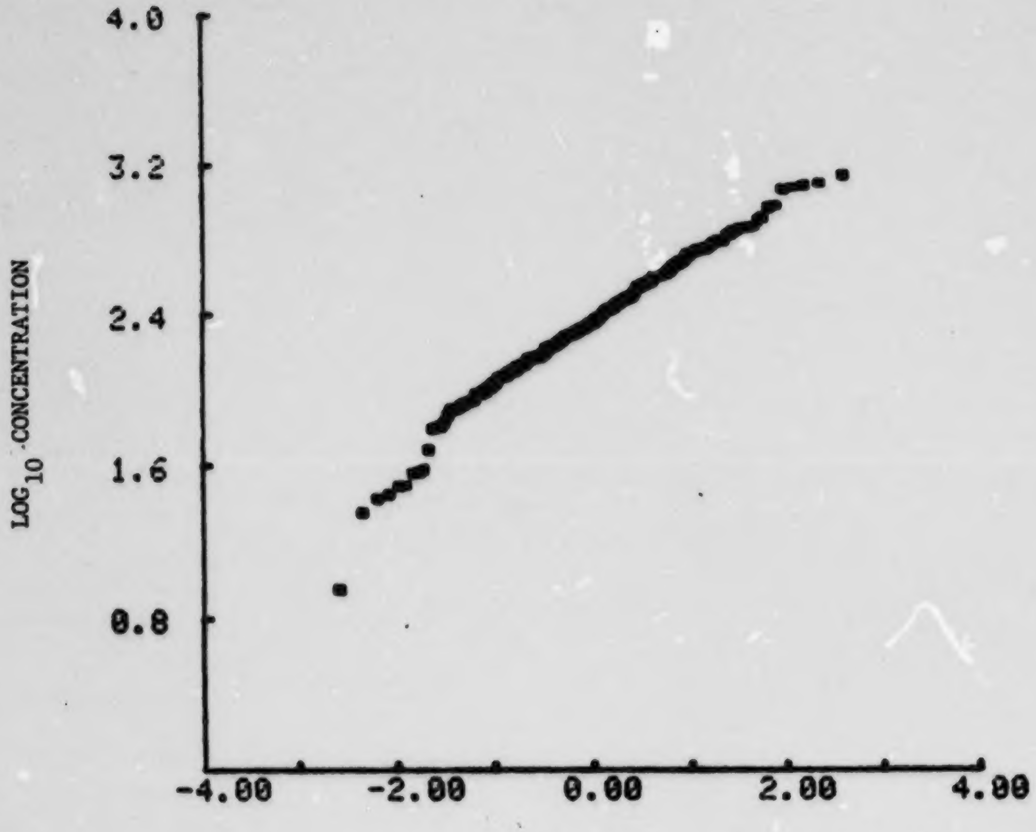

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Figure 2. BOD Concentration Data from Separate Sewer Systems

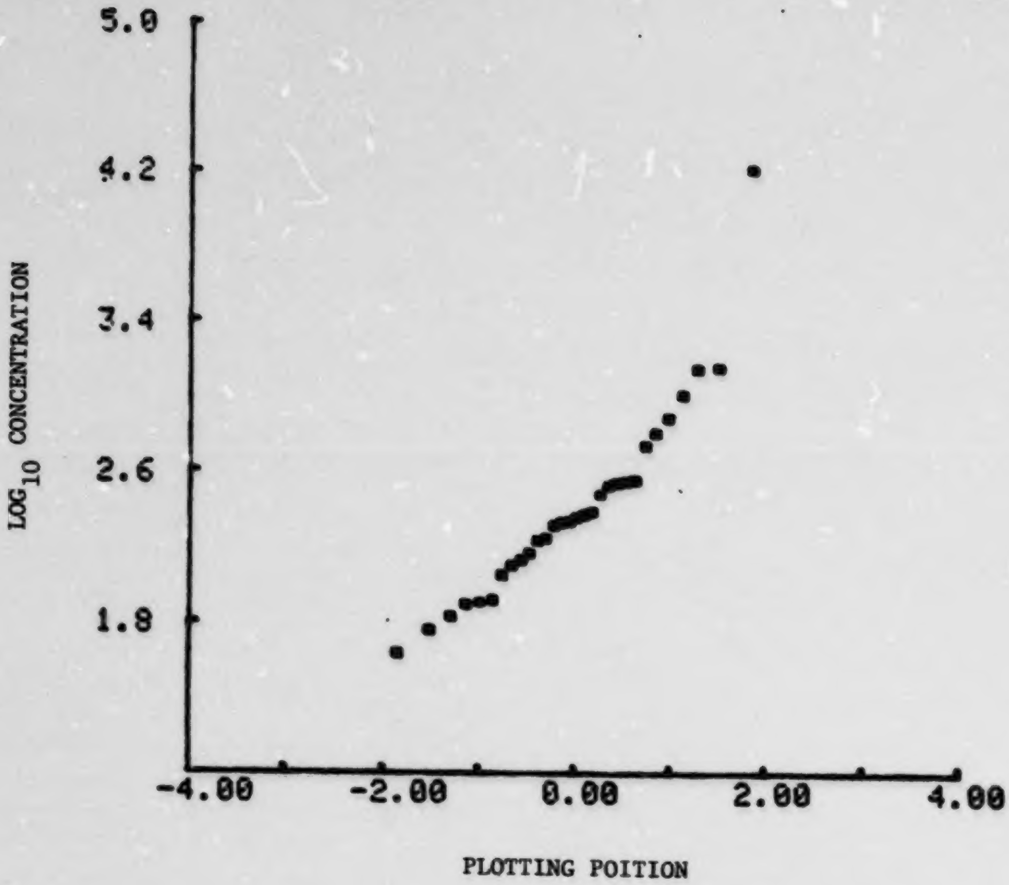


Figure 3. Lead Concentration Data from Combined Sewer Systems

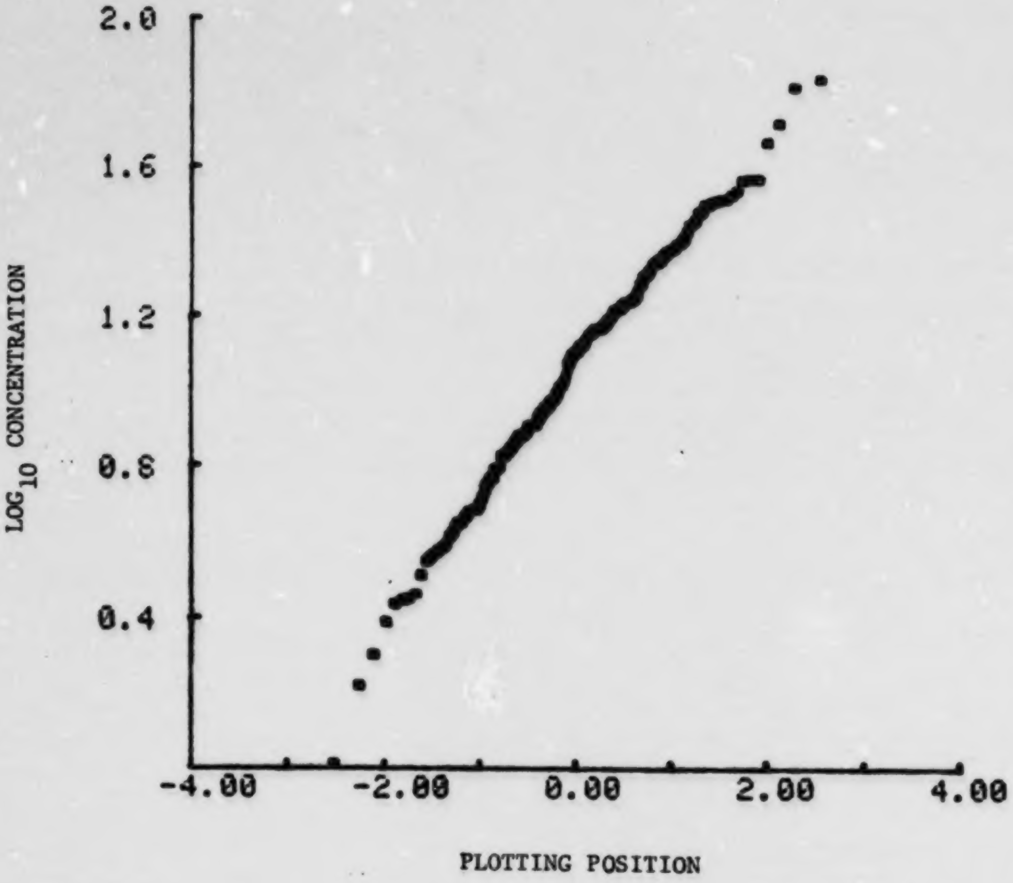


Figure 4. Lead Concentration Data from Separate Sewer Systems

TABLE 1

ℓn Median and Variance of Event Mean Concentrations for Urban Runoff Data

Contaminant	Median (ℓn)	Variance (of the ℓn)	No. Events
Ammonia (mg/ℓ)	-2.3	1.61	317
Nitrate (mg/ℓ)	-1.152	1.325	192
Total Nitrogen (mg/ℓ)	0.23	0.331	127
Ortho P (mg/ℓ)	-2.878	1.073	227
BOD_5 (mg/ℓ)	2.53	0.481	169
Suspended Solids (mg/ℓ)	5.066	2.239	264
Turbidity (JTU)	3.914	2.599	260
Zinc (µg/ℓ)	4.84	0.64	201
Arsenic (µg/ℓ)	3.79	0.015	21
Copper (µg/ℓ)	2.76	0.846	129
Lead (µg/ℓ)	5.4	0.64	206
Iron (µg/ℓ)	6.217	1.909	147
Cadmium (µg/ℓ)	0.921	0.477	146
Chromium (µg/ℓ)	2.99	0.48	127

TABLE 2

ℓn Median and Variance of Event Mean Concentrations for CSO

Contaminant	Median (ℓn)	Variance (of the ℓn)	No. Events
Ammonia (mg/ℓ)	0.691	0.889	91
Nitrate (mg/ℓ)	-0.231	1.605	77
Ortho P (mg/ℓ)	0	1.44	14
BOD_5 (mg/ℓ)	3.914	0.212	84
Suspended Solids (mg/ℓ)	5.181	0.649	98
Turbidity (JTU)	3.891	0.043	5
Zinc (μg/ℓ)	6.1	0.649	10
Arsenic (μg/ℓ)	3.914	0.005	3
Copper (μg/ℓ)	5.411	1.073	10
Lead (μg/ℓ)	5.526	1.603	30
Iron (μg/ℓ)	7.322	0.172	3
Cadmium (μg/ℓ)	2.763	1.323	8
Chromium (μg/ℓ)	5.066	1.061	8

SECTION 5

ANALYSIS OF IMPACTS IN RIVERS AND STREAMS

Figure 5 contains an idealized representation of a series of runoff discharges entering a stream. The discharges, which may be from CSO, urban runoff or a combination of the two, usually enter the stream at several locations, but can be aggregated into an equivalent discharge flow which enters the system at a single location. This consolidated discharge is used to estimate the water quality impacts from the sum of the individual discharges. The equivalent discharge flow (QR) is the sum of the individual discharges, and the equivalent concentration (CR) is the site mean concentration for the constituent of concern. If the mass discharged from individual sites is known, the average concentration is the total mass divided by total flow.

The receiving water concentration, which results from mixing the discharge with stream flow, is influenced by the upstream flow (QS) and the upstream concentration (CS) during a runoff event. Receiving water concentration (CO) is the resulting concentration after complete mixing of the runoff and stream flows, and should be interpreted as the average concentration just downstream of all of the discharges as shown in Figure 5.

The four variables that determine the stream concentration (CO) are:

1. Urban discharge flow (QR)
2. Urban discharge concentration (CR)
3. Stream flow (QS)
4. Stream Concentration (CS)

For an individual rainfall/runoff event, it is possible, in principle, to measure each of the variables independently. The average stream concentration (CO), during an individual storm event, could be calculated:

$$CO = \frac{(QR \cdot CR) + (QS \cdot CS)}{QR + QS} \qquad (1)$$

If a dilution factor, ϕ, is defined as:

$$\phi = \frac{QR}{QR + QS} \qquad (2)$$

CO may be defined in terms of ϕ by:

$$CO = [\phi \cdot CR] + [(1-\phi)CS] \qquad (3)$$

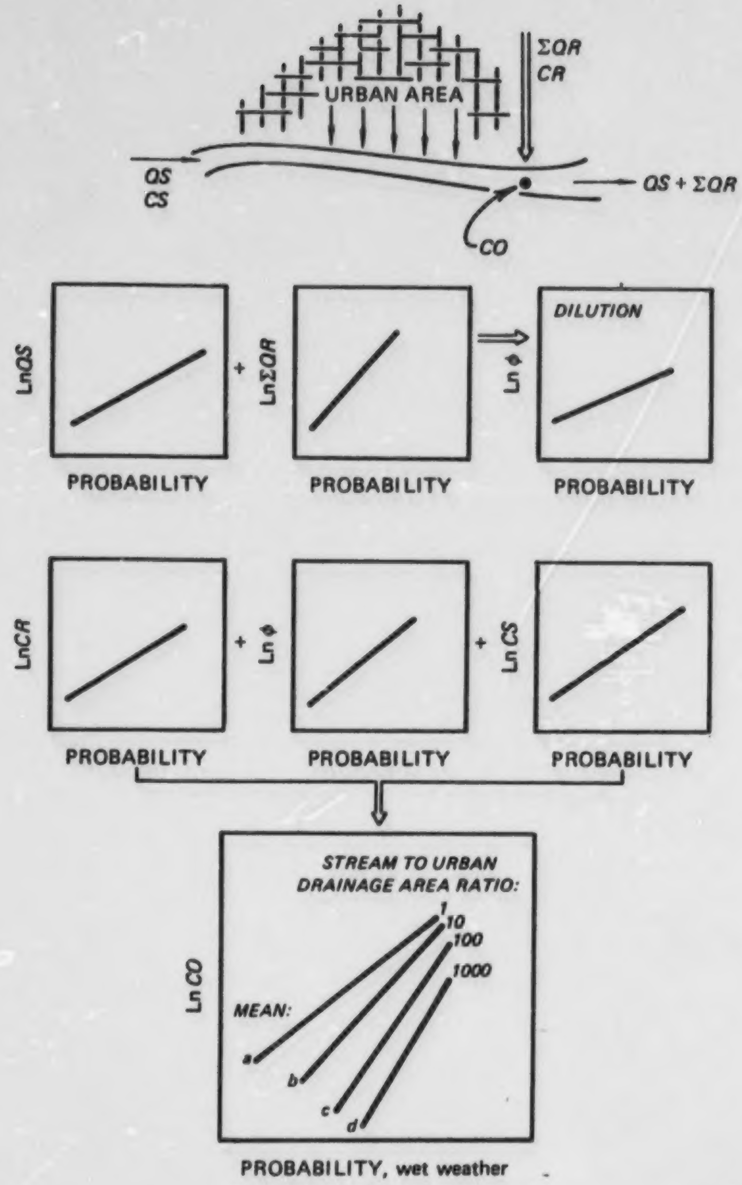

URBAN AREA
ΣQR
CR
QS
CS
QS + ΣQR
CO
LnQS
PROBABILITY
+
LnΣQR
PROBABILITY
Ln ϕ
DILUTION
PROBABILITY
LnCR
PROBABILITY
+
Ln ϕ
PROBABILITY
+
Ln CS
PROBABILITY
Ln CO
STREAM TO URBAN
DRAINAGE AREA RATIO:
1
10
100
1000
MEAN:
a
b
c
d
PROBABILITY, wet weather

Figure 5. Representation of Runoff Discharges Entering a Stream

The calculated value (CO) for an individual event could be compared to a water quality standard (CL), or to any other stream concentration which relates water quality to protection or impairment of water use. If the comparisons of CO and CL indicate that water quality is satisfactory, then it may be assumed that the individual event would not impair beneficial water usage. By contrast, if the comparison of CO and CL indicates that during an individual event receiving water concentrations of the constitutent in question would not protect beneficial usage, the relative contributions of runoff and upstream sources to the violation could be ascertained as follows:

$$CO = \underset{\text{Runoff}}{[\phi \cdot CR]} + \underset{\text{Upstream}}{[(1-\phi)CS]} \tag{4}$$

In principle, this procedure could be repeated for a large number of rainfall events and could also be applied during dry periods when urban runoff does not occur, and the runoff flow (QR) and concentration (CR) are zero. If this were done, the total percentage of the time (Pr_T) that CO caused impairment of beneficial water usage could be determined, and the relative contribution of rainfall/runoff events and upstream contributions could also be defined.

One measure of the relative impacts of runoff discharges which could be developed would be the percentage of the time during rainfall/runoff events (Pr_R) that CO caused impairment of water usage, compared with the percentage of the time during dry periods (Pr_D) that CO caused impairment. The relationship between these probabilities is:

$$Pr_T = \frac{D}{\Delta}(Pr_R) + \left(1 - \frac{D}{\Delta}\right)(Pr_D) \tag{5}$$

where Pr_T = overall probability of a violation, or that a selected concentration will occur (% of time, considering total time, including both wet and dry periods)

Pr_R = probability occurrence during wet periods

$\frac{D}{\Delta}$ = probability that it is raining. D = average duration of storms; Δ = average interval between storms

} % of total time (or probability) that violation occurs due to runoff events.

Pr_D = probability of violation occurring during dry periods (no runoff)

$1 - \frac{D}{\Delta}$ = probability that it is not raining, i.e., that there is no urban runoff

} % of total time (or probability) that violation occurs from causes other than urban runoff.

Decisions on the effectiveness of treatment of runoff discharges could be based on the total percent of the time the stream concentration (CO) was at undesirable levels and the relative contribution of runoff events.

A method has been developed by drawing resources, ideas and technology from this project, the National Urban Runoff Program (2), and the CSO analysis manual (3) project which, in essence, employs equations 1 to 5 and the statistical properties of the four variables to calculate the probability distribution of the instream concentration CO during runoff events and during periods without direct contributions from runoff.

This procedure is illustrated in Figure 5 for the periods influenced by wet weather events. An essential condition to the use of the approach is that each of the four variables which contributes to define instream receiving water quality can be adequately represented by ℓn-normal probability distributions. Examination of a reasonably broad cross-section of data indicates that ℓn-normal probability distributions can adequately represent discharges from the rainfall/runoff process, the concentration of contaminants in the discharge from urban overflows, and the daily flow record of many rivers and streams. Analysis of the probability distribution of daily stream flow records results in flow distribution curves which are employed, for example, in making hydroelectric power yield analyses. These curves can often be characterized by ℓn-normal probability distributions (4). The data base on instream concentrations of water quality variables, upstream of urban areas, such as heavy metals, BOD, D.O., etc. have not been extensively reviewed in this project. The author's experience, in past studies, indicates that ℓn-normal distributions of receiving water concentration data have provided adequate representations of the data at particular sites. The extent to which ℓn-normal distribution adequately represents upstream concentration data is unknown at this time.

The approach developed can be applied on a site specific basis. The approach can also be generalized and applied to a river system, region of the country or a series of locations which are characterized by similar rainfall and stream flow distributions. The ratio of the stream drainage area (above the urban area) to the drainage area of the urban area is the factor which allows this generalization. The year-one report of this project (5) discussed this ratio in detail pointing out the significance of the drainage area ratio and the role that is played by land use, and the volumetric runoff coefficient. Figure 5 illustrates the output which can be obtained through the generalized application of the approach employing the ratio of the stream to urban drainage area. The results presented in subsequent report sections employ this generalization. The example calculations discussed in this section consider a site specific application to illustrate the approach and to point out some of the limitations in the analysis framework.

The basic calculation procedure is summarized in table 3 and consists of elevan steps each of which are illustrated in the following example:

TABLE 3

1. Calculate the mean and variance of the logarithmic transforms of each of the four variables (QR, QS, CR, and CS).

2. Calculate the estimates of the mean and variance of the four variables. This calculation employs formulas that relate the arithmetic mean and variance to the mean and variance of the log transformations.

3. Calculate the mean and variance of the dilution factor, ϕ, employing the mean and variance of the logarithmic transforms of QR and QS. The calculation considers:

    a. Possible correlations between stream flow QS and runoff flow QR.

    b. Adjustments of the mean and variance of ϕ due to the upper bound of 1.0 on ϕ.

4. Calculate the arithmetic estimates of mean and variance of ϕ.

5. Calculate the mean and variance of CO using the estimates of the arithmetic mean and variance of CR, CS and ϕ.

6. Plot the log-normal cumulative probability distribution of stream concentration, CO. The mean and variance of the logarithmic transforms are used in developing the plot.

7. Define CL from a water quality standard or use other criteria to define a target concentration limit which will provide protection for beneficial water use.

8. Enter the log-normal cumulative probability plot for CO and read appropriate probability (P_R) for the CL selected.

9. Calculate the standard normal probability statistic, Z_D and Z_W, for both CS and CR, respectively, using the log mean and variance of the logarithmic transforms and ℓn CL.

10. Using tables or formulas, obtain the probabilities (PZ_W) and (PZ_D) for wet and dry periods, respectively.

11. Calculate P_R and P_0 for wet and dry periods employing (PZ_W) and (PZ_D) weighted by the expected percentage of time it is wet and dry.

EXAMPLE PROBLEM: STREAM/RIVER ANALYSIS URBAN DISCHARGE

FIRST ORDER REACTIVE SUBSTANCE

The notation employed in this example is defined according to the following random variables of concern in receiving water calculations:

CR = Concentration in stormwater discharge (μg/l or mg/l)

QR = Stormwater discharge flow rate (cfs or MGD)

CS = River concentration upstream of urban area (μg/l or mg/l)

QS = River flow upstream of urban area (cfs or MGD)

ϕ = Dilution in the receiving water

CO = Concentration in the stream at the outfall or below the stormwater discharges

CO(t)= Concentration in the stream at travel time "t" downstream

K = First order decay or removal rate

NOTE: Consistent concentration and flow units are required.

The basic equations in the stream used in the present example are:

$$\phi = \frac{QR}{QS + QR} \tag{6}$$

$$CO = (\phi \cdot CR) + [(1 - \phi)CS] \tag{7}$$

$$CO(t) = COe^{-tk} \tag{8}$$

Using the concentration in the stormwater discharge as an example of the notation, the following definitions are used:

CR = Random variable itself

μ_{CR} = Arithmetic average of CR

δ^2_{CR} = Variance of CR

ν_{CR} = Coefficient of variation of CR; $\nu_{CR} = \delta_{CR}/\mu_{CR}$

CR' = ℓn (Cr) base "e"

μ_{lnCR} = Arithmetic average of $CR' = \ell n\ (CR)$

CR = Median of CR; $CR = e^{\mu_{\ell nCR}}$

$\delta^2_{\ell nCR}$ = Variance of $CR' = \ell n\ (CR)$

Relationships between the arithmetic projection and the properties of a log-normally distributed variable are defined by:

$$\mu_{CR} = e^{\mu_{\ell nCR} + 1/2\delta_{\ell nCR}} = CR\sqrt{1 + \nu^2_{CR}} \qquad (9)$$

$$\nu_{CR} = e^{\delta^2_{\ell nCR}} - 1 \qquad (10)$$

$$\delta_{\ell CR} = \sqrt{\ell n(1 + \nu^2_{CR}} \qquad (11)$$

$$\delta_{CR} + \nu_{CR}\mu_{CR} \qquad (12)$$

For CR, which is distributed log-normally, and defining CR_α as the value of CR at the α percentile, i.e.:

$$Pr \{CR \leq CR\} = \alpha$$

Then $$CR_\alpha = e^{\mu_{\ell nCR} + Z_\alpha \delta_{\ell nCR}}$$

where: Z_α is the standard normal percentiles given in Table 4 as Z'.

Data for the concentration of zinc in runoff are illustrated on Figure 6 which represents a log-normal plot of Zn concentration and cumulative frequency as a percent. The line representing the data on this plot may be drawn by eye. The median value at the 50% probability is read off the curve as 118 μg/l Zn.

Employing the definition previously presented:

$CR = 118$ μg/l

$\mu_{1nCr} = \ell n\ (118) = 4.77$

The value of 84% (84.13%) probability is also read from the straight line as CR @ 84% = 280. The variance δ^2_{1nCR} of $CR' = \ell n\ (CR)$ is calculated as follows:

CR @ 84% = 280
$\ell n(CR\ @\ 84\%) = 5.63$
$\delta_{1nCR} = \ell n\ (CR\ @\ 84\%) - \mu_{1nCR} = 5.63 - 4.77 = 0.86$

$\delta^2_{1nCR} = 0.74$

employing equation (10)

$$\nu^2_{CR} = e^{\delta^2_{1nCR}} - 1 = e^{0.74} - 1 = 2.096 - 1 = 1.096 = 1.1$$

$$\nu_{CR} = 1.05$$

TABLE 4

CUMULATIVE STANDARD NORMAL DISTRIBUTION

Probabilities for Values of z

Z"	P(z < z')	z'	P(z < z')	z'	P(z < z')	z'	P(z < z')
-4.0	.0000	-2.0	.0228	0	.5000	2.0	.9772
-3.9	.0000	-1.9	.0287	.1	.5398	2.1	.9821
-3.8	.0001	-1.8	.0359	.2	.5793	2.2	.9861
-3.7	.0001	-1.7	.0446	.3	.6179	2.3	.9893
-3.6	.0002	-1.6	.0548	.4	.6554	2.4	.9918
-3.5	.0002	-1.5	.0668	.5	.6915	2.5	.9938
-3.4	.0003	-1.4	.0808	.6	.7257	2.6	.9953
-3.3	.0005	-1.3	.0968	.7	.7580	2.7	.9965
-3.2	.0007	-1.2	.1151	.8	.7881	2.8	.9974
-3.1	.0010	-1.1	.1357	.9	.8159	2.9	.9981
-3.0	.0013	-1.0	.1587	1.0	.8413	3.0	.9987
-2.9	.0019	- .9	.1841	1.1	.8643	3.1	.9990
-2.8	.0026	- .8	.2119	1.2	.8849	3.2	.9993
-2.7	.0035	- .7	.2420	1.3	.9032	3.3	.9995
-2.6	.0047	- .6	.2743	1.4	.9192	3.4	.9997
-2.5	.0062	- .5	.3085	1.5	.9332	3.5	.9998
-2.4	.0082	- .4	.3446	1.6	.9452	3.6	.9998
-2.3	.0107	- .3	.3821	1.7	.9554	3.7	.9999
-2.2	.0139	- .2	.4207	1.8	.9641	3.8	.9999
-2.1	.0179	- .1	.4602	1.9	.9713	3.9	1.0000

Values of z for Selected Probabilities

z'	P(z < z')	z'	P(z < z')
-3.090	.001	0.6745	.750
-2.576	.005	1.282	.900
-2.326	.010	1.645	.950
-1.960	.025	1.960	.975
-1.645	.050	2.326	.990
-1.282	.100	2.576	.995
-0.6745	.250	3.090	.999

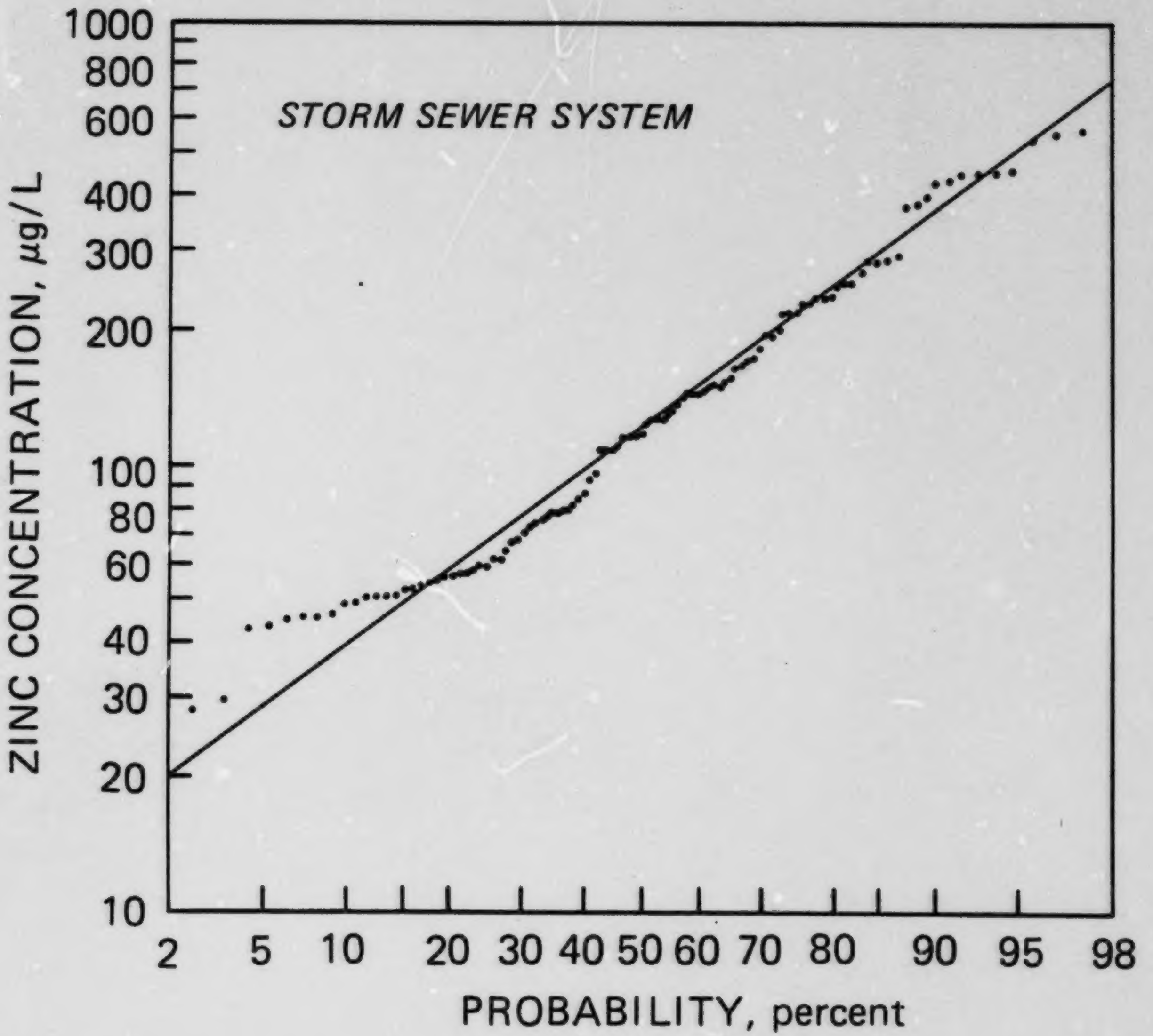


Figure 6. Zinc Concentration Data from Separate Storm Sewer Systems

and employing equation (9)

$$\mu_{CR} = 118\sqrt{1 + 1.1} = 118 \times 1.45 = 171.1 \ \mu g/l.$$

employing equation (12)

$$\delta_{CR} = \nu_{CR}\mu_{CR} = 1.05 \times 171 = 179.66$$

$$\delta^2_{CR} = 179.66^2 = 32275.9$$

The identical calculation techniques can be employed for each of the log-normally distributed variables whose distributions are presented on Figures 7, 8 and 9. Table 5 may be constructed from the results of these calculations.

TABLE 5. LOG NORMAL AND ARITHMETIC PROPERTIES OF THE FOUR VARIABLES OF CONCERN IN STREAM CALCULATIONS

Variable	$\tilde{\text{Median}}$	μ_{ln}	δ^2_{ln}	μ	ν^2	δ^2
CR	118 μg/l	4.77	0.74	171.1	1.1	32275.
QR	78.98 cfs	4.37	1.02	112.25	1.77	22288
CS	20 μg/l	3.0	1.12	35	2.06	2540.
QS	220 cfs	5.39	0.85	336.5	1.32	152365.

The dilution ϕ may be estimated for any percentile "α" employing equation (13):

$$\phi_\alpha = \frac{\tilde{QR}}{\tilde{QR} + \tilde{QS}\, e^{Z_\alpha \delta_{lnSR}}} \qquad (13)$$

where the variables are defined as before and in addition:

$$\delta_{lnSR} = \sqrt{\delta^2_{QS} + \delta^2_{QR} - 2\,\rho_{lnSR}\,\delta_{QS}\delta_{QR}} \qquad (14)$$

$$\rho_{lnSR} = \frac{1}{N}\sum_{1}^{N} \frac{(\ell n(QS) - \mu_{lnQS})}{\delta_{QS}} \frac{(\ell n(QR) - \mu_{lnQR})}{\delta_{QR}} \qquad (15)$$

ρ_{lnSR} = correlation coefficient between runoff and stream flow.

N = rainfall events 1, 2, 3 . . . n

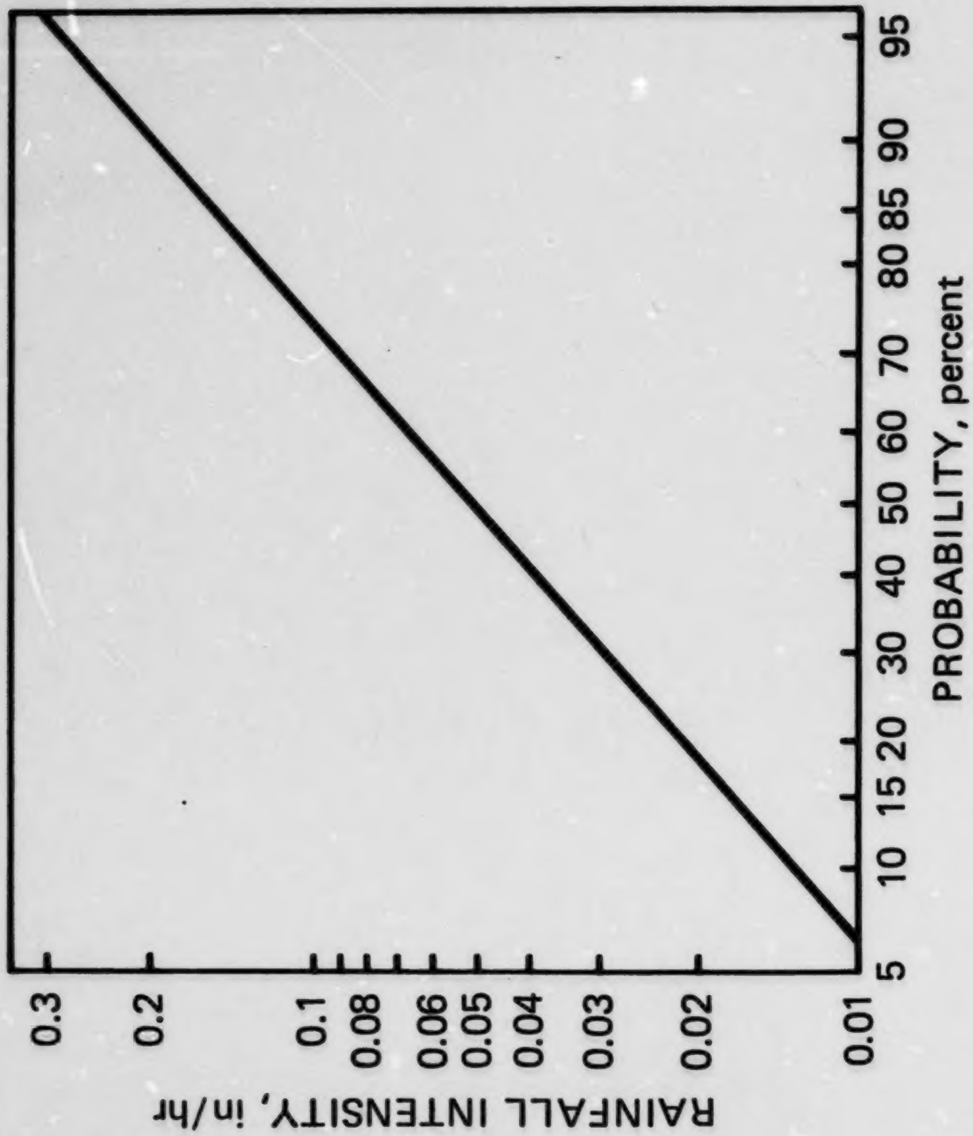


Figure 7. Log-Probability Distribution of Rainfall Intensity

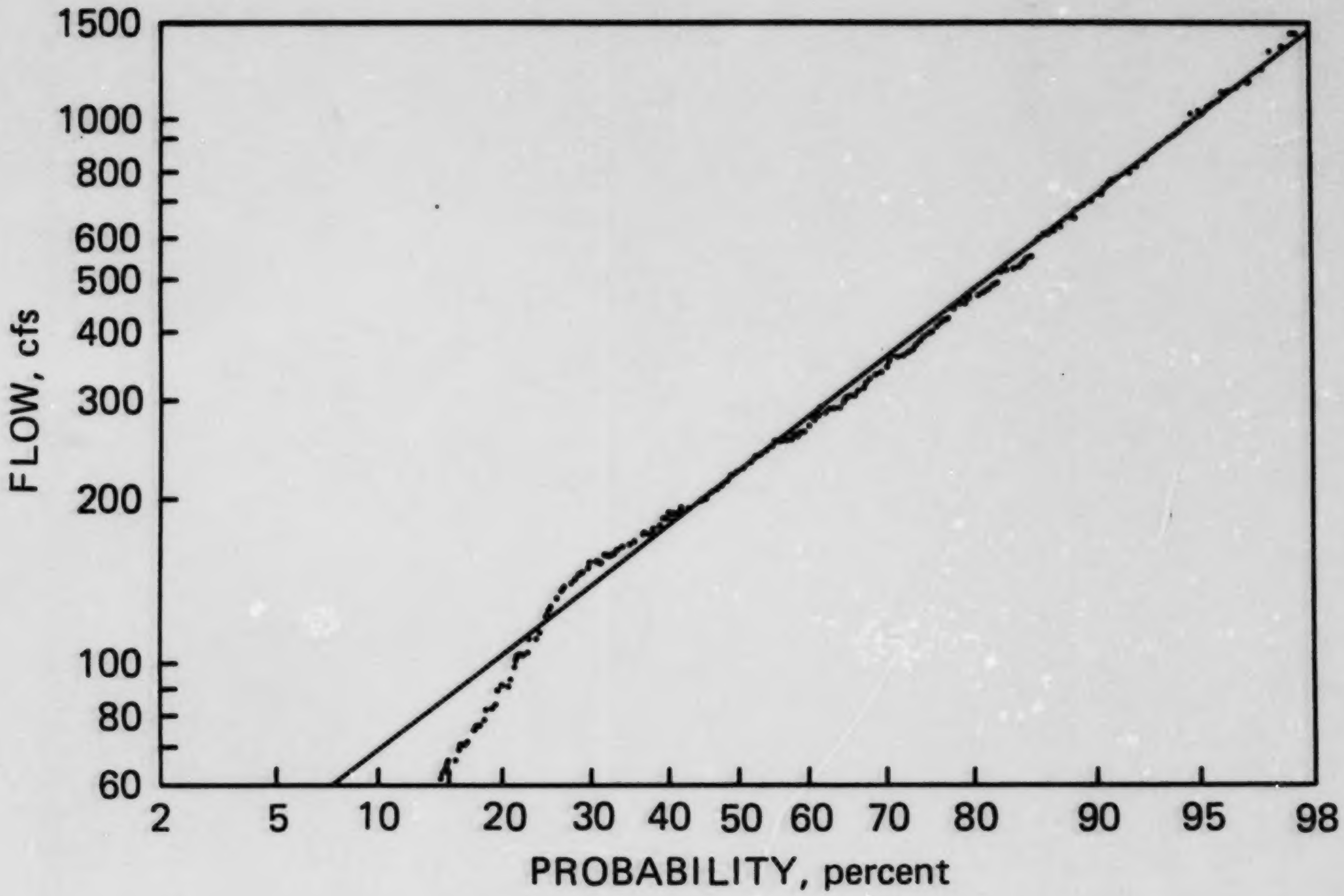


Figure 8. Log-Probability Distribution of Flow at Ramapo River near Mahway October 1971-September 1972

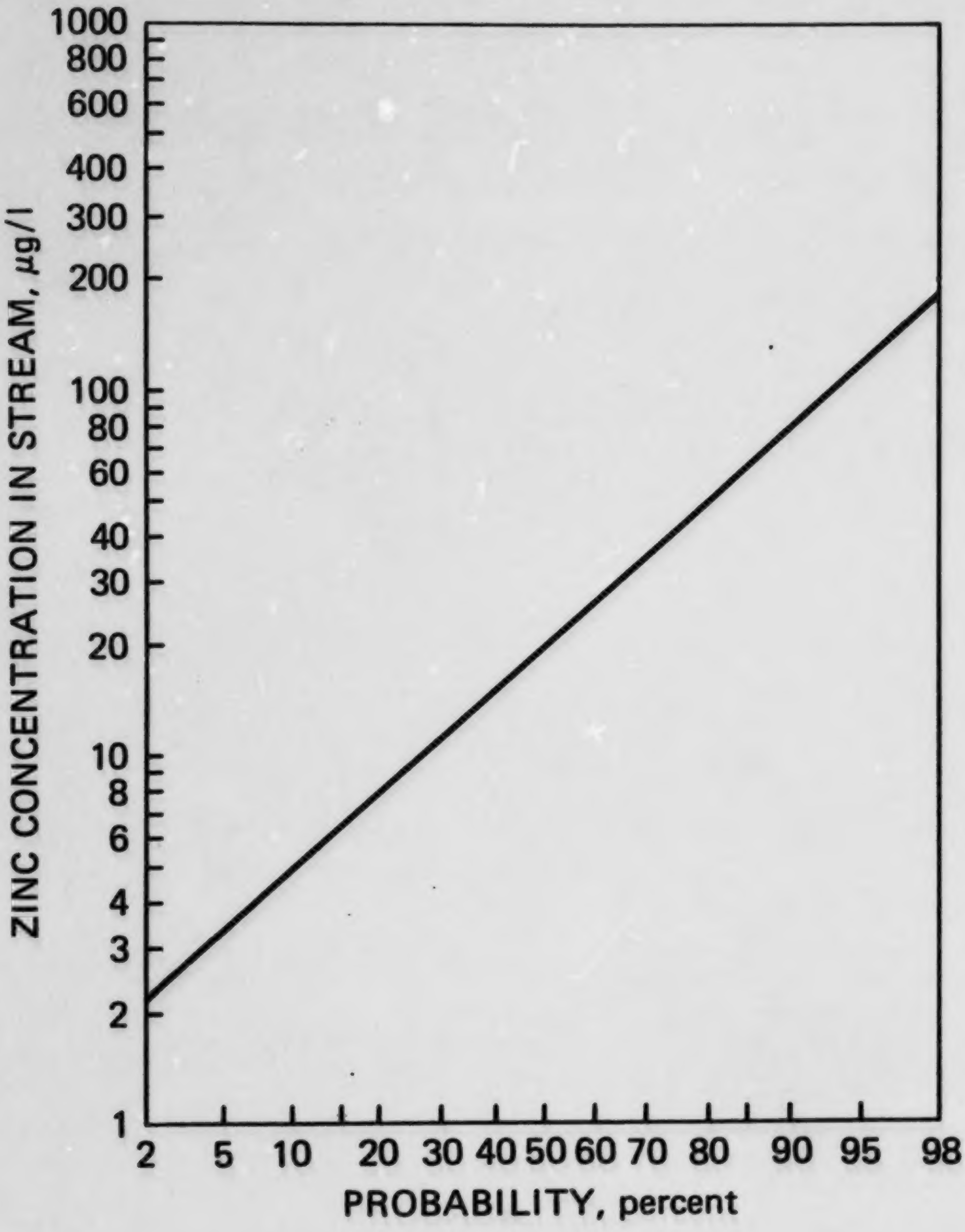


Figure 9. Zinc Concentration in the River Above the Stormwater Discharge Location

The stream flow QS may be correlated to the runoff flow QR in some basins since rainfall patterns which cross the drainage areas of streams above the urban area will tend to produce increases in stream flow as well as CSO runoff. Therefore, in some systems, large storm overflows will tend to be associated with larger stream flows. The correlation coefficient ρ_{OR} accounts for this tendency.

Calculating $\delta_{\ln SR}$ using equation (14) assuming $\rho_{\ln SR} = 0.5$

$$\delta_{\ln SR} = \sqrt{0.85 + 1.02 - 2 \times .5 \times .92 \times 1.01} = 0.97$$

From Table 4, $Z = 1.645$ for $\alpha = 95\%$ and $Z = -1.645$ for $\alpha = 5\%$ using equation (13) to calculate ϕ_α yields:

$$\phi_{95} = \frac{78.98}{78.98 + 220^{+1.645 \times .97}} = \frac{78.98}{78.98 + 220 \times 4.93} = 0.068$$

$$\phi_5 = \frac{78.98}{78.98 + 220^{-1.645 \times .97}} = \frac{78.98}{78.98 + 220 \times .203} = 0.64$$

The calculation above will insure that estimates of stream dilution falls between 0 and 1.0 somewhat beyond the 90% interval bounded by 5 and 95 percent. This calculation procedure is an approximation of the actual distribution of the dilution factor ϕ. This particular approximation should be replaced by the actual distribution for decisions which are addressing probabilities in excess of 95%. Solutions to the actual distribution have recently been developed(2) and are substantially more complex. The calculation procedure presented could be employed as a screening analysis while the more complex computations would be appropriate for site specific projects or projects with large resources. The median dilution may be estimated by equation (16):

$$\mu_{\ell n\phi} = \frac{1}{2}\left[\ell n\,(\phi_{95}) + \ell n\,(\phi_5)\right] = \frac{1}{2}\left[-2.69 + (-.446)\right] \qquad (16)$$

$$\mu_{\ell n\phi} = -1.57;\ \tilde{\phi} = e^{-1.57} = 0.21$$

The variance of dilution "$\delta_{\ell n\phi}$" may be calculated by equation (17):

$$\delta_{\ell n\phi} = \frac{1}{2 x Z_\alpha}\left[\ell n(\phi_{95}) - \ell n(\phi_5)\right] = 0.304\ [-2.69 + .45] = .68 \qquad (17)$$

The arithmetic mean can be calculated employing equation (9):

$$\mu_\phi = e^{\mu_{\ell n\phi} + \frac{1}{2}\delta^2_{\ell n\phi}} = e^{-1.57 + \frac{-.68^2}{2}} = 0.262$$

The arithmetic variance, δ_{ϕ}, is defined by equation (18):

$$[e^{\delta^2_{\ell n\phi}}-1]^{\frac{1}{2}}\ \frac{1}{2} = 0.262\ [e^{.46}-1]^{\frac{1}{2}}\ \frac{1}{2} = 0.20 \qquad (18)$$

The properties of dilution are:

$\bar{\phi} = 0.21$

$\mu_{\ell n\phi} = -1.57$

$\delta_{\ell n\phi} = -.68$

$\mu_{\phi} = 0.26$

$\delta_{\phi} = 0.20$

These may be employed to construct a probability plot as shown in Figure 10 for available dilution.

The arithmetic mean of the receiving water contaminant concentration just downstream from the CSO discharges can be calculated employing equation (19) which is comparable to equation (7):

$$\mu_{CO} = (\mu_{CR} \cdot \mu_{\phi}) + \mu_{CS}\ (1 - \mu_{\phi}) \qquad (19)$$

$$\mu_{CO} = 171.1 \times .262 + 35\ (1 - .262) = 70.6\ \mu g/\ell$$

The arithmetic variance δ^2_{CT} can be calculated employing equation (20):

$$\delta^2_{CO} = \delta^2_{\phi}\ (\mu_{CR} - \mu_{CS})^2 + \delta^2_{CR}\ (\delta^2_{\phi} + \mu^2_{\phi}) + \delta^2_{CS}[\delta^2_{\phi} + (1 - \mu_{\phi})^2] \qquad (20)$$

$$= .04\ (171.1-35)^2 + 32276\ (.04+.26^2) + 2540\ [.04 + (1-.26)^2]$$

$$= .04 \times 18523 + 32276 \times .11 + 2540 \times .58$$

$\delta^2_{CO} = 5764$

$\delta_{CO} = 75.9$

$\nu^2_{CR} = 5764 / 70.6^2 \quad 1.155$ (from equation 12)

$\mu_{\ell nCO} = \ell n\ (70.65/\sqrt{1+1.155}) = \ell n\ (48.1) = 3.87$ (from eq. 9 and

$\mu_{\ell nCO} = \ell n\ (CO)$)

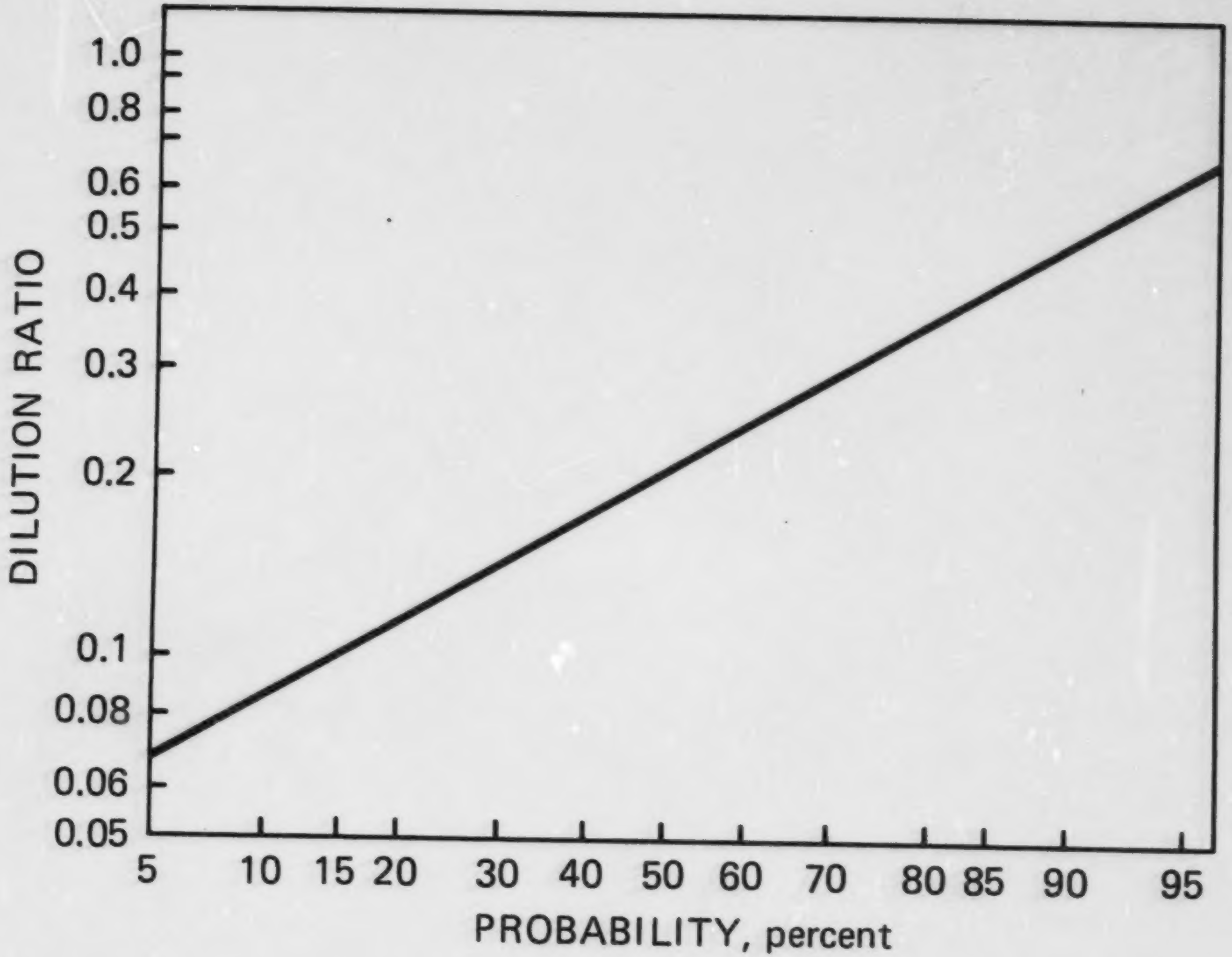


Figure 10. Log Probability Distribution of Dilution Ratio Based on a Stream and Runoff Flow Correlation Coefficient of 0.5

$$CO = e^{3.87} = 47.9 \ \mu g/\ell$$

$$\delta_{CO} = \sqrt{\ell n\ (1+1.155)} = 0.88 \text{ (from equation 11)}$$

The properties of CO are:

$CO = 47.9$ g/ℓ
$\mu_{\ell nCO} = 3.87$
$\delta_{\ell nCO} = 0.88$
$\mu_{CO} = 70.6$
$\delta_{CO} = 75.9$

These properties were employed to develop the probability curve presented in Figure 11 for CO the instream concentration of zinc just below the discharge from the urban area. From the figure or from equation (21):

$$CO_\alpha = e^{\mu_{\ell nCO} + Z_\alpha \cdot \delta_{\ell nCO}} \qquad (21)$$

$$CO_{95} = e^{3.87+1.645x.88} = 203.9 \ \mu g/\ell$$

$$CO_{90} = e^{3.87+1.282x.88} = 148.1 \ \mu g/\ell$$

$$CO_{50} = e^{3.87} = 47.9 \ \mu g/\ell$$

The figure and/or equation 21, together with the properties of CO listed define the variation of the concentration of zinc in the stream below the discharge during periods of rainfall and runoff. Note that CO is the stream concentration during runoff events which are assumed to be controlled by the runoff contamination rather than the upstream contamination. The method for calculation of the percent exceedance of a target concentration CL a distance further downstream with first order decay (by settling, for example) can be obtained for periods influenced by runoff events and for periods which are not influenced by runoff events, the following procedure is employed:

Define a target concentration CL = 50 μg/ℓ at t = x/u = 1 day travel time with a removal rate of k = 0.1/day.

$\mu_{\ell nCO} = 3.87$
$\delta_{\ell nCO} = .88$
$\mu_{\ell nCS} = 3.0$
$\delta_{\ell nCS} = 1.06$

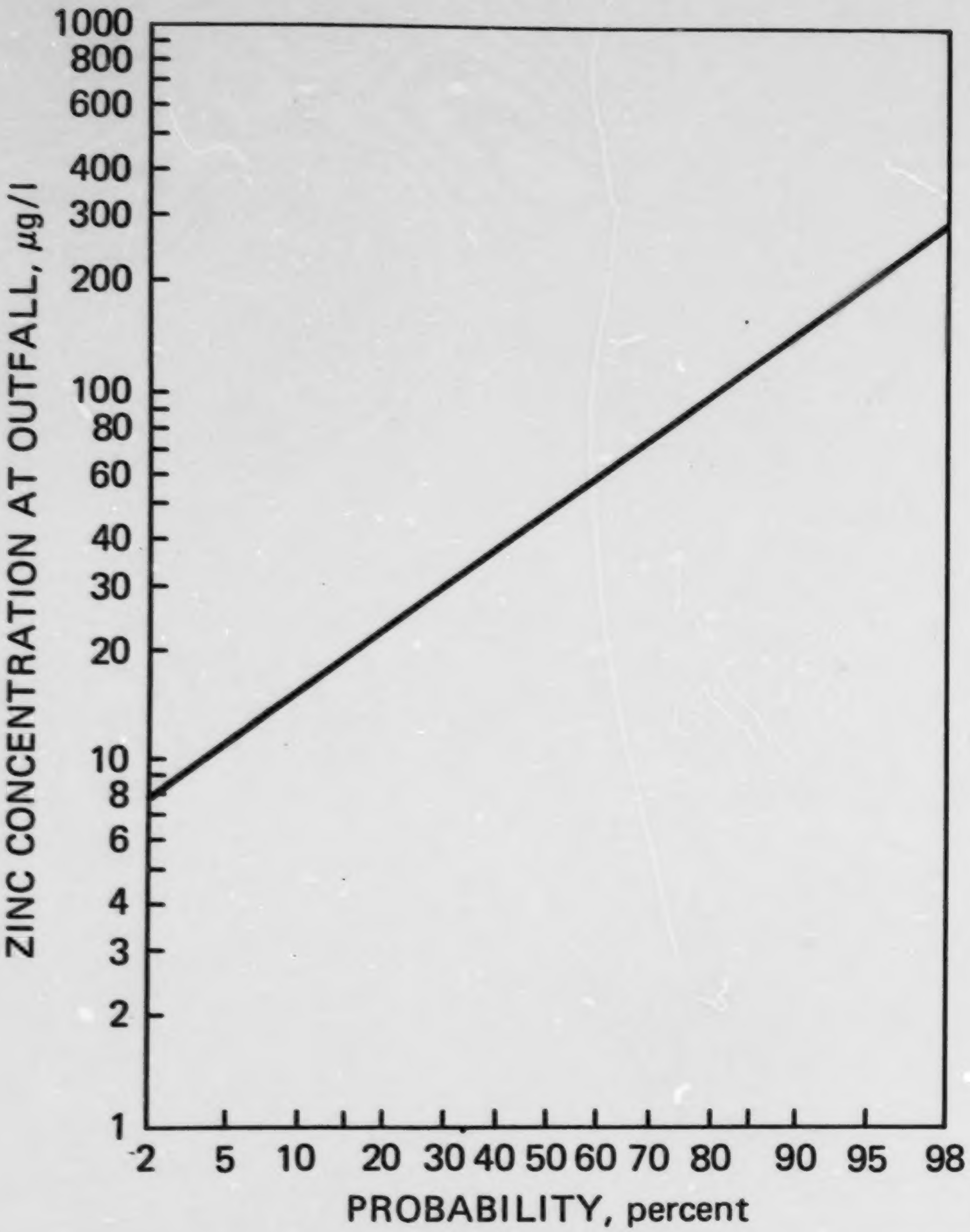


Figure 11. Zinc Concentration in the Stream Near Overflow

The probability that the stream concentration CO(t) at a travel time "t" will exceed CL is defined by equation (22).

$$\Pr\{CO(t) > CL\} = \frac{D}{\Delta}\, P(Z_{CLtR}) + \left(1 - \frac{D}{\Delta}\right) P(Z_{CLt}) \tag{22}$$

where D = duration of runoff events (estimate of the arithmetic mean)

Δ = interval between runoff events (estimate of the arithmetic mean)

t = travel time

$CO(t)$ = concentration at travel time t

CL = target concentration

$P(Z_{CLtR})$ = Probability of CO(t) exceeding CL at time of travel "t" during periods controlled by runoff

$P(Z_{CLt})$ = Probability of CO(t) exceeding CL at time of travel "t" during periods with no runoff contribution

$$P(Z_{CLtR}) = \frac{1}{2} - \frac{1}{2}\sqrt{1 - e^{-2Z_{CLtR}/\pi}} \tag{23}$$

$$Z_{CLtR} = \frac{\ell n(CL) + kt - \mu_{\ell nCO}}{\delta_{\ell nCO}} \tag{24}$$

Equations (23) and (24) may be employed to calculate the probability of exceedance during periods influenced by runoff. Equations (25) and (26) may be employed to calculate the probability of exceedance during periods which are not influenced by runoff.

$$P(Z_{CLt}) = \frac{1}{2} - \frac{1}{2}\sqrt{1 - e^{-2Z_{CLt}/\pi}} \tag{25}$$

$$Z_{CLt} = \frac{\ell n(CL) + kt - \mu_{\ell nCS}}{\delta_{\ell nCS}} \tag{26}$$

Substituting into equations (24) and (26):

$$Z_{CLtR} = \frac{\ell n(50) + 1 \times 0.1 - 3.87}{.88} = 0.159 \text{ (eq. 24)}$$

$$Z_{CLt} = \frac{\ell n(50) + 1 \times 0.1 - 3.0}{1.06} = 0.953 \text{ (eq. 26)}$$

Continuing the numerical substitutions into equations 23 and 25 yields:

$$P(Z_{CLtR}) = \frac{1}{2} - \frac{1}{2}\sqrt{1 - e^{-2 \times .159^2/\pi}} = .4368 \text{ (eq. 23)}$$

$$P(Z_{CLt}) = \frac{1}{2} - \frac{1}{2}\sqrt{1 - e^{-2 \times .953^2/\pi}} = 0.169 \text{ (eq. 25)}$$

From rainfall statistics:

D = 7.65 hrs. (duration of rain)

Δ = 80.8 hrs. (interval between storms)

Substituting into equation (22) yields:

$$\Pr\{CO(t) > CL\} = \frac{7.65}{80.8} \times .4368 + (1 - \frac{7.65}{80.8}) \times .169$$

$$= .041 + .153 = .194$$

The total probability of the stream concentration exceeding 50 μg/ℓ at one day travel time is 19.4 percent. This is composed of 4.1 percent due to runoff and 15.3 percent due to the upstream concentrations in the stream.

If Z_{CLtR} or Z_{CLt} is calculated to be a negative quantity, then $[1 - P(Z_{CLtR})]$ or $[1 - P(Z_{CLt})]$ is substituted into eq. (22) instead of the actual values calculated from eq. (23) and eq. (25), respectively.

Application And Limitations:

The procedure outlined can be employed to:

1. Estimate water quality violations (as a probability of violating a criteria) at various times of travel downstream from a discharge considering 1st order decay of the contaminant in the discharge.
2. Calculate an estimate the probability distribution caused by individual waste sources.
3. Calculate the effects of any discharge (intermittent or continuous) so long as the concentration and flow of the discharge can be adequately represented by a ℓn-normal probability distribution. (Most point source discharges meet this criteria)
4. The procedure can be employed to estimate the total probability distribution of the instream concentration considering all time periods.

The limitations of the procedure are as defined below:

1. This is essentially a dilution calculation. Data to verify the results of this calculation has not been collected. The results of the calculation, therefore, should be viewed in the context of an unverified dilution calculation. Verification of the calculation procedure is particularly significant in view of the assumptions described for the probability distribution of dilution ϕ.
2. The calculation procedure which includes the reaction rate and travel time should also be viewed as unverified. Further, it should be noted that the location (stream mile point) associated with any travel time (say 1 day) other than $t = o$ will vary with the total downstream flow. Therefore, the method does not provide an indication of the physical location of the calculated probability distribution when a travel time other than $t = o$ is considered.
3. The approach does not consider:
   a. Tributary inputs downstream of the discharge
   b. dispersion
   c. variable reaction rates in space, or time.
4. The method requires and employs ℓn-normal representations of the significant variables. Ln-normal distributions of some of the significant variables at specific sites may not adequately represent the tails of the distributions (low or high probability events). When this is the case the tails of the calculated distributions may not adequately represent the system.

In summary, the analysis framework is capable of estimating the probability distribution of instream water quality. These calculations coupled with information on water quality criteria can be used to define situations where CSO and/or urban runoff loads could result in water quality problems. The method does have limitations and each application of the method should be examined in the context of these limitations.

Dissolved Oxygen Analysis for Rivers and Streams:

The statistical procedure discussed previously can be employed to analyze the ultimate oxygen demand (UOD) associated with the discharges from urban areas. Additional computations are required to relate the UOD to dissolved oxygen deficit and subsequently to target concentrations of dissolved oxygen which are associated with impairment of beneficial usage.

The dissolved oxygen deficit at the critical location can be defined by considering constant reaction rates (K_a,K_d) and equation (27)(6),

$$D_c = L_o \, \Phi^{\frac{\Phi}{1-\Phi}} \tag{27}$$

where D_c = dissolved oxygen deficit at the critical location associated with the urban discharge

L_o = UOD at the discharge location (CSO or urban runoff but assuming no upstream concentration)

Φ = purification ratio, K_a/K_d

K_a = reaeration coefficient

K_d = deoxygenation coefficient

If there is a significant upstream flow, L_o is reduced by dilution using equation (28). It is assumed there is no accompanying deficit with this additional flow.

$$L_o = \frac{L_r Q_o}{Q_o + Q_s} \qquad (28)$$

where Q_o = discharge flow

L_r = discharge UOD concentration

Q_s = stream flow

The procedure entails calculating D_c for a specific UOD discharge, dilution ratio and purification ratio that occurs at the site in question. This critical deficit is compared to the stream dissolved oxygen saturation value, C_s, the water quality criteria to be maintained, C_{limit}, and deficits contributed by other sources, D_x, by equation (29).

$$C_s - D_x - D_c \geq C_{limit} \qquad (29)$$

Equation (29) should be satisfied if water quality criteria are to be met. In this context D_c may be viewed as the dissolved oxygen deficit which may be allocated to the urban discharge.

This calculation can be used to evaluate the probable effect that runoff resulting from a specific storm event will have on the stream into which it discharges. For example, suppose the stream and storm event are subject to the following conditions:

L_r = 50 mg/ℓ

Dilution ratio = 2

Φ = 5

C_s = 9.0 mg/ℓ

C_{limit} = 4.0 mg/ℓ

D_x = 2.0 mg/ℓ

From eq. (28):

$$L_o = \frac{50 \times Q_o}{Q_o + 2Q_o} = 16.667 \text{ mg}/\ell$$

Substituting into eq. (27):

$$D_c = 16.667 \ (5)^{\frac{5}{1-5}} = 2.23$$

Testing in eq. (29):

$$9.0 - 2.0 - 2.23 \geq 4.0$$

$$4.77 \geq 4.0$$

Therefore, the calculation indicates that the D.O. standard is not violated due to this storm.

The procedure can be followed in reverse to analyze a specific discharge site. The calculation procedure indicated above can be rearranged to estimate the allocatable UOD during storm events. This is accomplished by solving equation (29) for the allowable deficit D_c. This deficit is used in equation (27) to calculate the allocatable ultimate oxygen demand L_o which is the oxygen demand equivalent of the target concentration CL discussed in the previous section.

Figure 12 relates the allocatable ultimate oxygen demand "L_o" to the allowable dissolved oxygen deficit "D_c" for various values of the purification ratio Φ.

In summary, the ultimate oxygen demand which may be discharged to streams or rivers during storm events without impairment of beneficial usage will vary depending upon the deoxygenation rate "K_d" and the reaeration rate "K_a" of an individual water body. The procedure for dissolved oxygen analysis is subject to the same ranges of application and limitation as discussed previously. In addition the assumptions of constant and definable deoxygenation and reaeration rates provide additional significant restrictions. It is suggested that a consistent set of maximum deoxygenation rate and minimum reaeration rates be employed in the analysis. In this case, the calculated percentage of the time that criteria are violated should be viewed as an upper bound estimate for the critical period. The analysis procedure should, where possible, be directed to critical seasonal periods. This is in contrast to the general application of the statistical approach; for example toxics; where an annual time scale can be used.

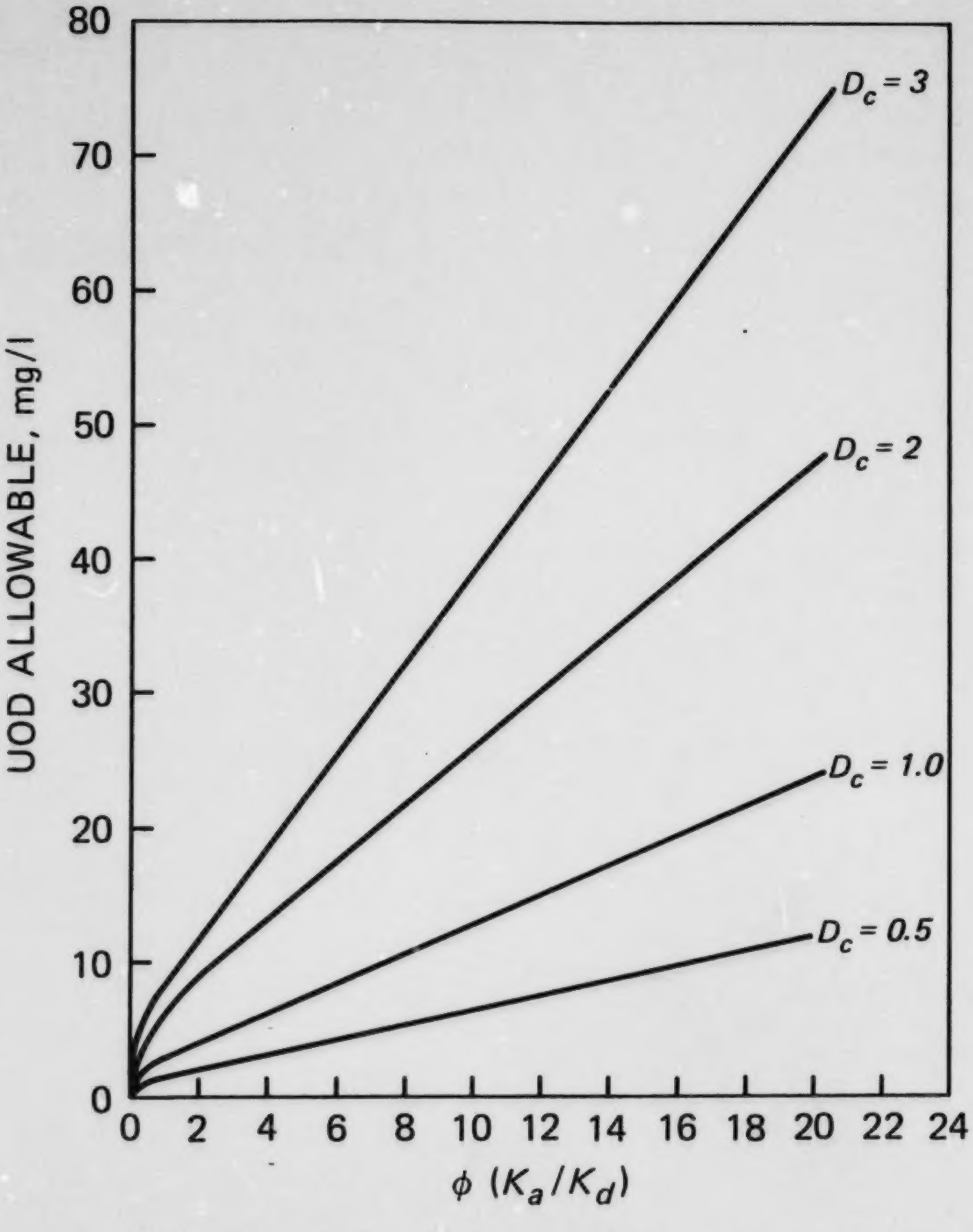


Figure 12. UOD Concentration Allowable in the Discharge for Varying Purification Ratios (ϕ) and Critical Deficits (D_c)

SECTION 6

BIOLOGICAL RESPONSES TO TIME VARIABLE CONCENTRATIONS

Conceptual View of Experimental Results:

A substantial body of experimental data is available which examines the biological responses, as measured by mortality, of aquatic organisms exposed to constant concentrations of a contaminant or mixture of contaminants (7)(8). This bioassay data which employs lethality as the criterion of toxicity is usually reported in terms of the Lethal Concentration (LC) with a numerical value representing the percentage of organisms killed (LC_{50}). Time is also included in the expression of results such that a 96 hr-LC_{50} is that concentration of a contaminant which is lethal to 50 percent of the test organisms at 96 hrs. of exposure.

Data obtained from lethal bioassays usually consists of tabulated values of the percent mortality observed at various times of exposure for several concentration levels of a contaminant. The laboratory procedures (9) employed, in essence, consist of exposure of a random sample of organisms to a constant concentration of contaminant in an experiment chamber with periodic observations of organism mortality. Assuming mortality occurs, there are two types of responses possible, i.e., all the organisms die at one time or, the organisms die at different times. The latter is usually observed in lethal bioassays and indicates that there is a distribution of sensitivity of the organism population to the contaminant at the concentration being tested. In this context the percent mortality can be viewed as a measure of population sensitivity where there is a distribution of sensitivities in the total population. One method of representing this data (10) is to plot the percentage of organisms killed vs. time for constant concentrations. This representation of the data is consistent with the experimental procedures utilized and is essentially a plot that represents the observations obtained from each experiment chamber.

Lethal bioassay data can often be represented on a log probability plot. As illustrated on figure 13 the log of time can be plotted against the percent mortality on the probability scale with a separate straight line for each concentration. This representation is consistent with the essence of the experimental procedures and the concept of a distribution of organism sensitivity. Organisms at a given sensitivity level in each of the constant concentration experiments can be assumed to have common or similar characteristics with respect to the effects of the chemical being tested. By contrast, the organisms which represent two different sensitivity levels at the same exposure concentration have differing characteristics. For example, the organisms represented by points A_1 and A_2 represent the 5 percent sensitivity level for the experiments at 1 mg/ℓ and 0.40 mg/ℓ of zinc and therefore

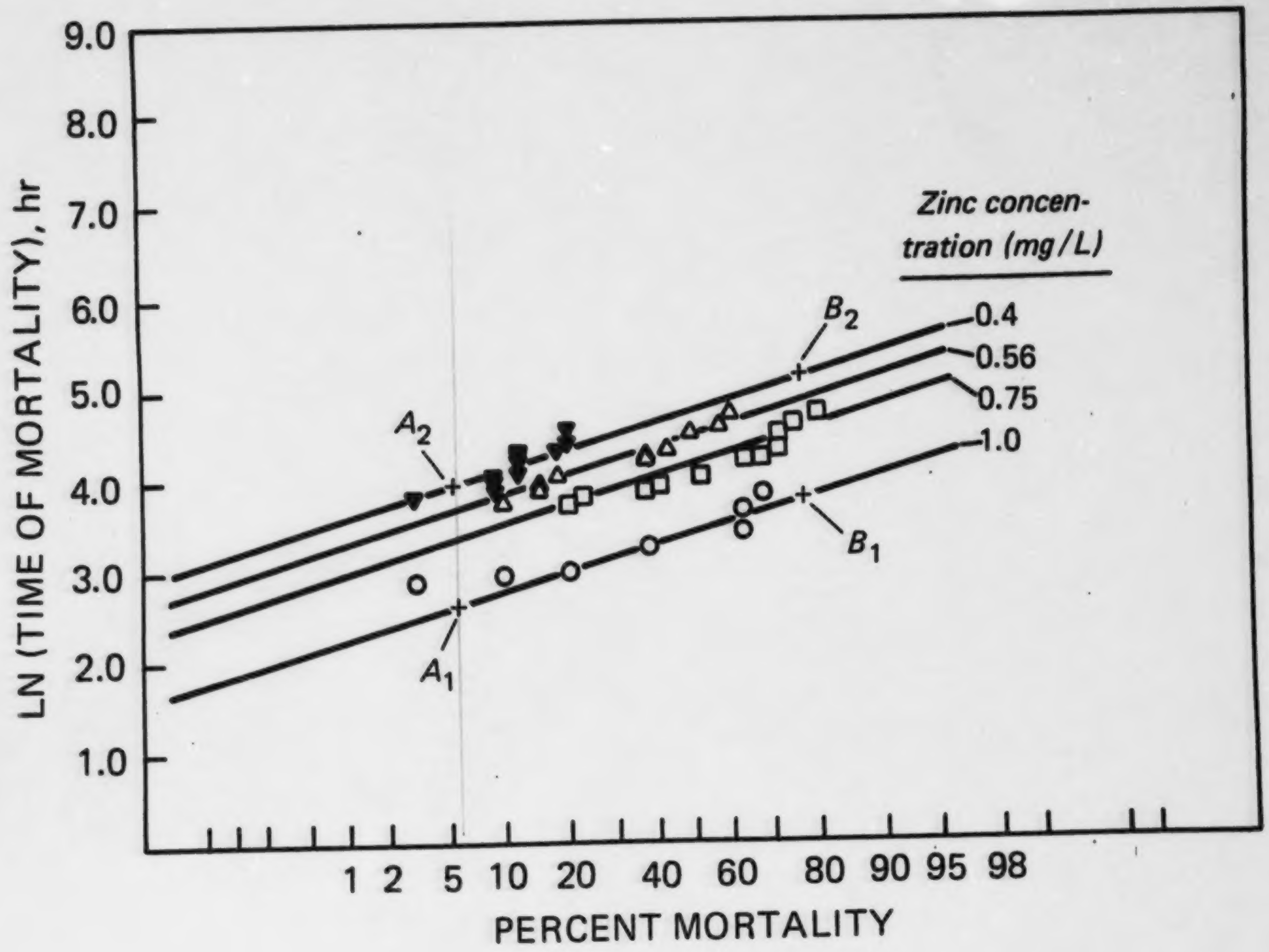


Figure 13. Illustration of Lethal Bioassay Data

have common response characteristics. By contrast, the organisms represented by points A_1 and B_1 have different response characteristics since they represent two different sensitivity levels within the total population. Employing similar reasoning, the organisms represented by points B_1 and B_2 have common characteristics. This characterization of organism response and sensitivity appears to be somewhat different than that which has been used in some recent studies (11),(12).

The data presented in figure 13 is, as indicated, for experimental results which can be represented by a log normal distribution. The conceptualization of a distribution in the sensitivity of the population is equally applicable for any distribution of sensitivities, even for data sets where the underlying distribution cannot be identified. Further, the conceptual analysis and subsequent numerical evaluations employ lethality as the measure of effect. The approach can also be employed utilizing other measures of effects such as reproductive success, swimming ability, etc. which can be substituted for mortality.

The characteristics that represent various sensitivity levels in the population can include the rate of entry of the contaminant into the organism, the rate of depuration or detoxification within the organism, and/or the concentration in the organism which results in mortality.

The mechanisms by which toxics are taken up by aquatic organisms are quite complex, undoubtedly containing a series of transfers of the toxic from the point of entry, in the body, to the specific organ or site where functional impairment results in death. Similarly, detoxification in the sense of mortality effects, could include return of a toxicant to the water environment, destruction of the toxic, and organic or inorganic complexation or binding of the toxicant at sites which do not cause mortality. The body burden of a toxic at the start of an exposure can influence overall organism sensitivity and could be a result of direct uptake from the water as well as the uptake associated with the food web and ingestion of contaminated food sources.

Methods of Analysis:

The totality of processes and factors which interact to determine the time of mortality of an individual organism after exposure to a toxic are complex. The approach taken in this project is to identify relatively simple mechanistic approximations for the actual processes which occur and ultimately result in mortality.

A first approximation considers uptake of the toxic accompanied by detoxification as shown in figure 14. The rate of change of toxicant concentration in the organism is defined by equation (30).

$$\frac{dc_N}{dt} = k_u \cdot c_w - k_R \cdot c_N \qquad (30)$$

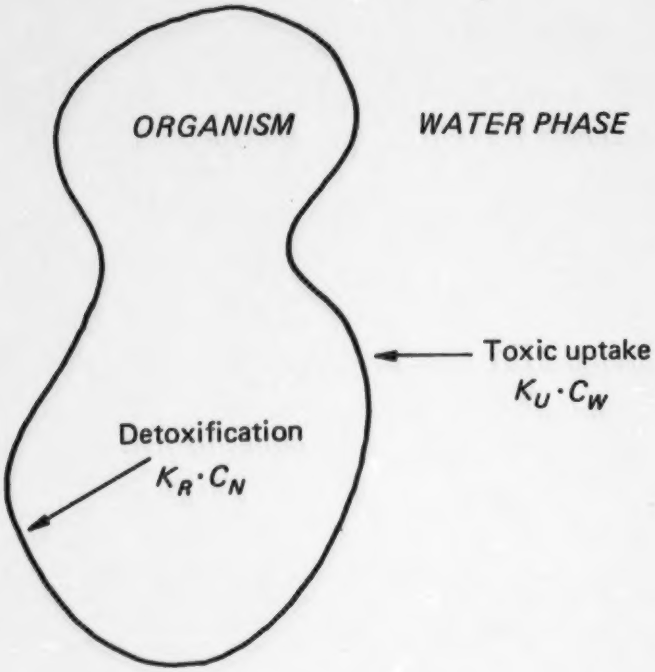

ORGANISM
WATER PHASE
Toxic uptake
$K_U \cdot C_W$
Detoxification
$K_R \cdot C_N$

Figure 14. Simplified Representation of Toxic Uptake and Detoxification Which Could Result in Organism Mortality

where:

c_N = organism related concentration of toxic $(\frac{\text{mass toxic}}{\text{mass organism}})$

c_w = concentration of toxic in water $(\frac{\text{mass toxic}}{\text{volume water}})$(constant overtime)

k_R = detoxification rate (1/time)

t = time of exposure (time)

k_u = toxic uptake rate $(\frac{\text{vol. water}}{\text{mass organism} \cdot \text{time}})$

Integration of equation (30) between the limits of:

$$@\ t = 0 \qquad c_N = 0 \qquad \text{and}$$

$$@\ t = t_D \qquad c_N = c_D$$

and separating c_w and t_D yields:

$$\frac{1}{c_w} = (\frac{k_u}{c_D}) \frac{1}{k_R} [1 - e^{-k_R t_D}] \tag{31}$$

where c_D = organism related concentration "c_D" at death.
t_D = time of organism death.

Equation (31) contains two separable coefficients which can be evaluated from lethal bioassay data obtained for exposures at constant toxic concentrations. These two coefficients are "k_R" and "$\frac{k_u}{c_D}$" which represent the common characteristics associated with data obtained for the same sensitivity level of the test population. Figures 15 and 16 contain data for several experiments (11) in which guppies were exposed to constant concentrations of zinc and cyanide. The points represent experimental data while the solid lines are calculated employing equation (31) and constant values of the two coefficients "k_R" and "$\frac{k_u}{c_D}$" for each sensitivity level as shown on the figures.

The coefficient "k_R" represents the rate of detoxification over the period of exposure. By contrast the value of the coefficient "k_u/c_D" is associated with the time of organism mortality since it is evaluated when the organism related concentration "c_N" is equal to "c_D" at the time of organism mortality. An analysis of the units of the quantity "c_D/k_u" in-

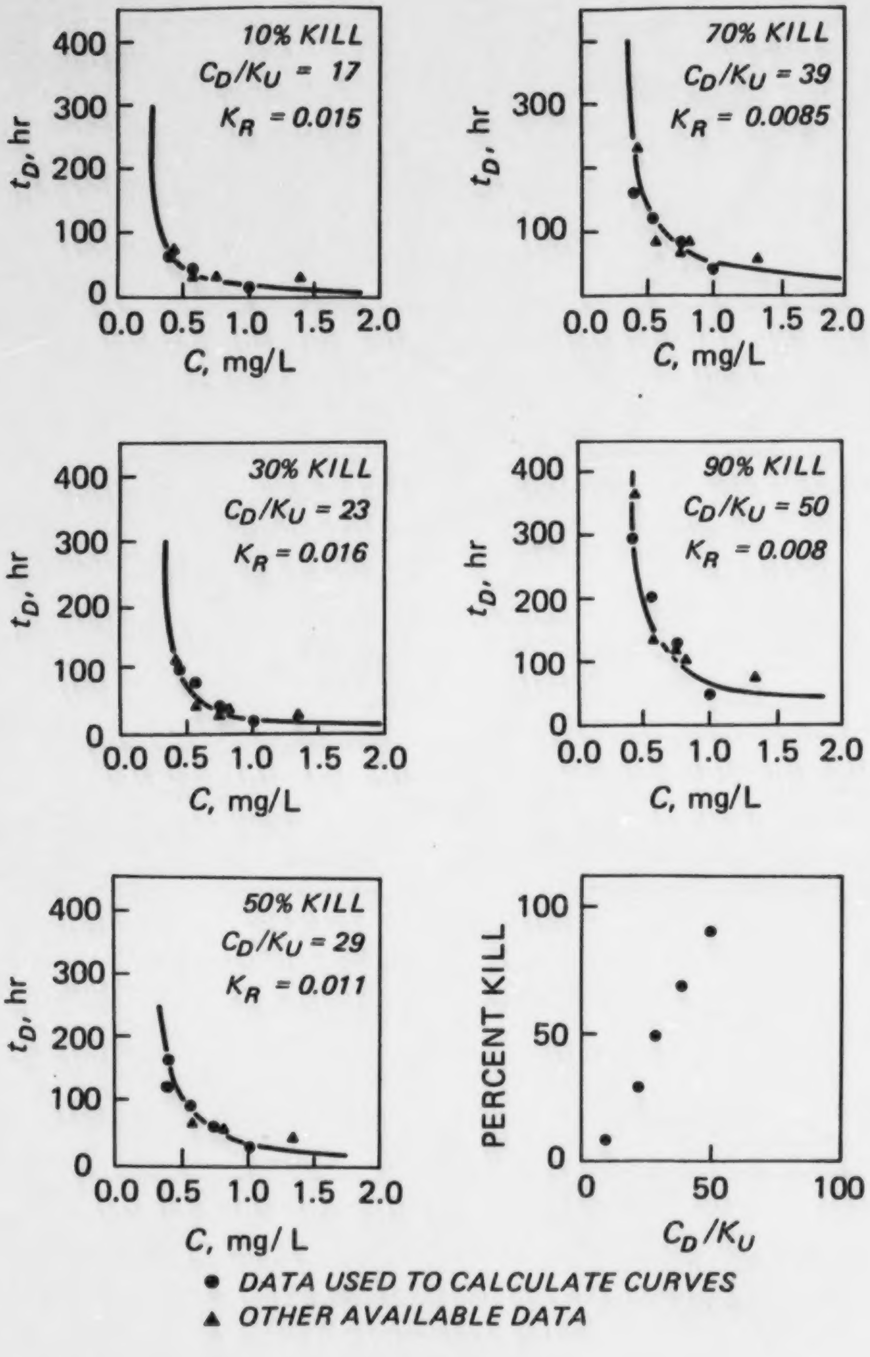


Figure 15. The Effects of Zinc on Guppies

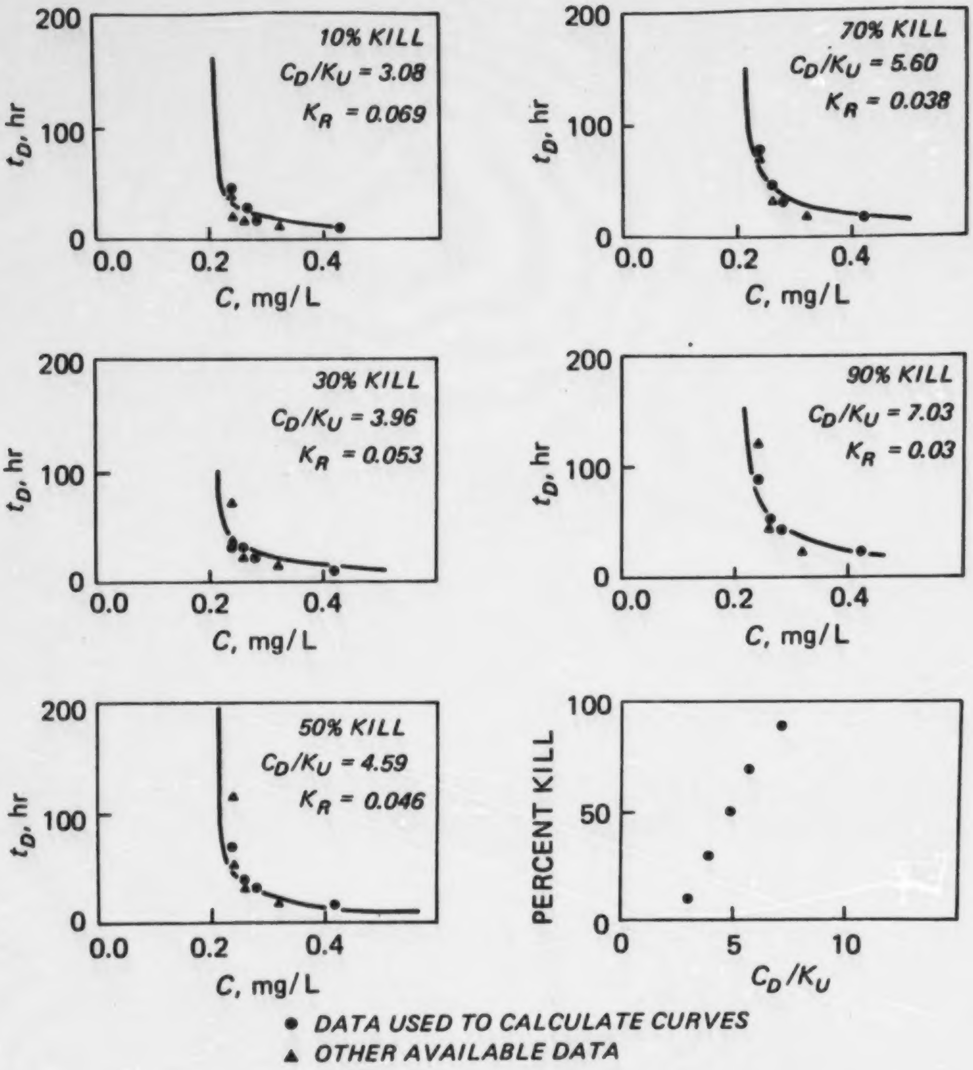


Figure 16. The Effects of Cyanide on Guppies

dicates that this coefficient has units of toxic mass multiplied by time and divided by water volume. This is a measure of an equivalent product of exposure time and water concentration. Therefore the coefficient "c_D/k_u" can be viewed as the equivalent toxic dose required for mortality and has a physical/biological interpretation. This coefficient shall be referred to as the equivalent mortality dose in the remainder of the report.

Equation (31) can be rearranged to yield equation (32).

$$\frac{c_D}{k_u} = \frac{c_w}{k_R} [1-e^{-k_R t_D}] \tag{32}$$

where the value of the "equivalent mortality dose", "c_D/k_u" is reached only at the time of organism mortality. For all exposure times "t" less than that required for mortality equation (32) becomes equation (33).

$$\frac{c_{Nt}}{k_u} = \frac{c_w}{k_R} [1-e^{-k_R t}] \tag{33}$$

where the variables are as defined previously and c_{Nt}/k_u is the calculated "equivalent dose" after exposure t. Considering the case where an organism is exposed to a series of concentrations as shown in fig. 17, equation (33) can be applied to time period (1). The equation for analysis of time period (2) may be obtained by integration of equation (30) between the limits of $t = o$ $c_N = c_{Nt}$ and at $t = t_1$ $c_N = c_{Nt1}$ which results in equation (34).

$$\frac{c_{Nt1}}{k_u} = \overbrace{\frac{c_{wi}}{k_R} [1-e^{-k_R t_1}]}^{\text{Buildup}} + \overbrace{\frac{c_{Nt}}{k_u} e^{-k_R t_1}}^{\text{Decay}} \tag{34}$$

The value of c_{Nt}/k_u is obtained from equation (33) for time period (1) and represents an initial condition which decays away in equation (34). Therefore the terms in equation (34) represent a buildup due to the current exposure concentration level "c_{wi}" and a decay term for "equivalent dose" at the beginning of the current exposure. Equation (34) may be continually reapplied for any sequence of exposures to constant concentrations (such as shown for time period (3)) where the term "c_{Nt}/k_u" is the ending value of "equivalent dose" from the previous exposure calculation. For water concentrations "c_{wi}" which vary over time, averaging of concentrations over short time periods with application of equation (34) for each time interval can be used. Organism mortality for a particular level of sensitivity occurs when the value of "c_{Nt1}/k_u" equals "c_D/k_u" (i.e. "equivalent dose" = "equivalent mortality dose"). The calculations for a particular sensitivity level after mortality has occurred are meaningless and can be terminated.

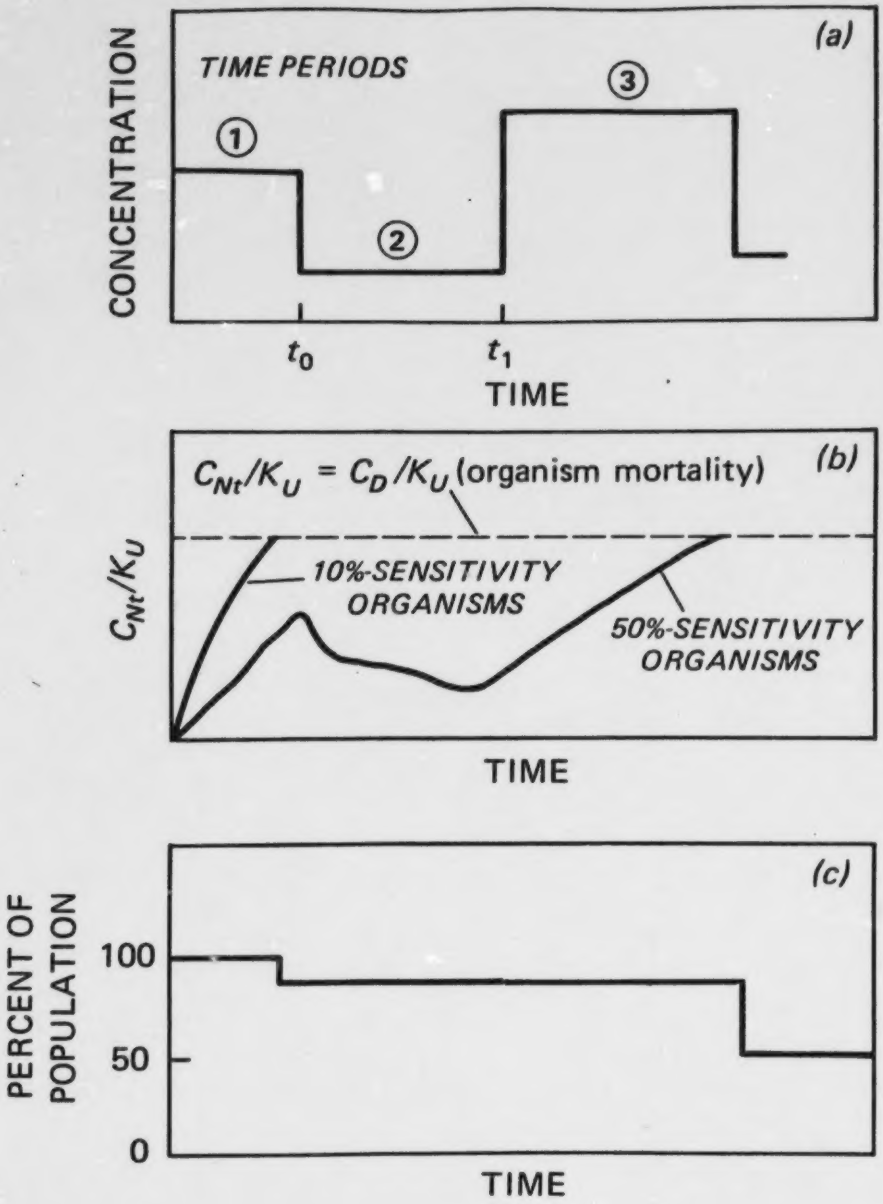


Figure 17. Illustration of the Use and Output of the Calculation Framework

Figure 17 illustrates a sequence of results that would be associated with exposure to variable concentrations. Since the detoxification rate varies for each sensitivity level there would be a series of "c_{Nt1}/k_u" vs. time curves (figure 17b) for each level of sensitivity considered. The composite effect on total population is shown in figure 17c.

As an aside, the analysis framework would tend to yield a "Darwinian" survival of the fittest with the less sensitive organisms tending to survive.

Test of the Method:

A data base has been found (13) which provides information on trout mortality due to exposure to constant concentrations of zinc and also contains data on trout mortality from time variable exposure to zinc. The analysis framework discussed to this point was employed to analyze the available data. Trout mortality data were available for constant exposure concentrations of 4 and 6 mg/ℓ in water with a hardness of ∿ 320 mg/ℓ as $CaCO_3$. These data are shown by the points of figure 18. The lines represent approximations of the observed data. Figure 19 contains the results of the time variable experiments for two and four hour alternate exposures to 2 mg/ℓ and 6 mg/ℓ of zinc. The solid lines were the calculated mortalities associated with the time variable exposures. These solid lines were calculated using only the constant exposure data at 4 and 6 mg/ℓ.

Additional data sets of the type required to test the analysis framework were generated by Seebode (14) who employed the Golden Shiner (Notemigonus crysoleucas), measured zinc concentrations, and in New York City tap water. The constant concentration results presented in figure 20 were analyzed to define "k_R" and "c_D/k_u" for various sensitivity levels. These coefficients were then employed to calculate expected mortality from exposure to time variable concentrations. In one experiment the concentrations were alternated each four hours at 2 and 10 mg/ℓ zinc. A second experiment employed concentrations at 4 and 10 mg/ℓ zinc which alternated each four hours. Figures 21 and 22 present the observed data as points and the calculated line represents the predicted mortality time using the "k_R" and "c_D/k_u" values from the constant concentration experiments presented in figure 20.

The results shown in figures 19, 21 and 22 suggest that the analysis framework presented is capable of calculating mortality for organisms exposed to time variable concentrations using mortality data from exposure to constant concentrations.

Family of Formulations:

Most but not all lethal bioassay data can be adequately represented by equations (33) and (34). It has been found that some data can be represented by substituting "$c_w^{1/2}$" or "$c_w^{1/4}$" for "c_w" in these equations. Therefore the comparable analysis framework for these data would employ equations (35)

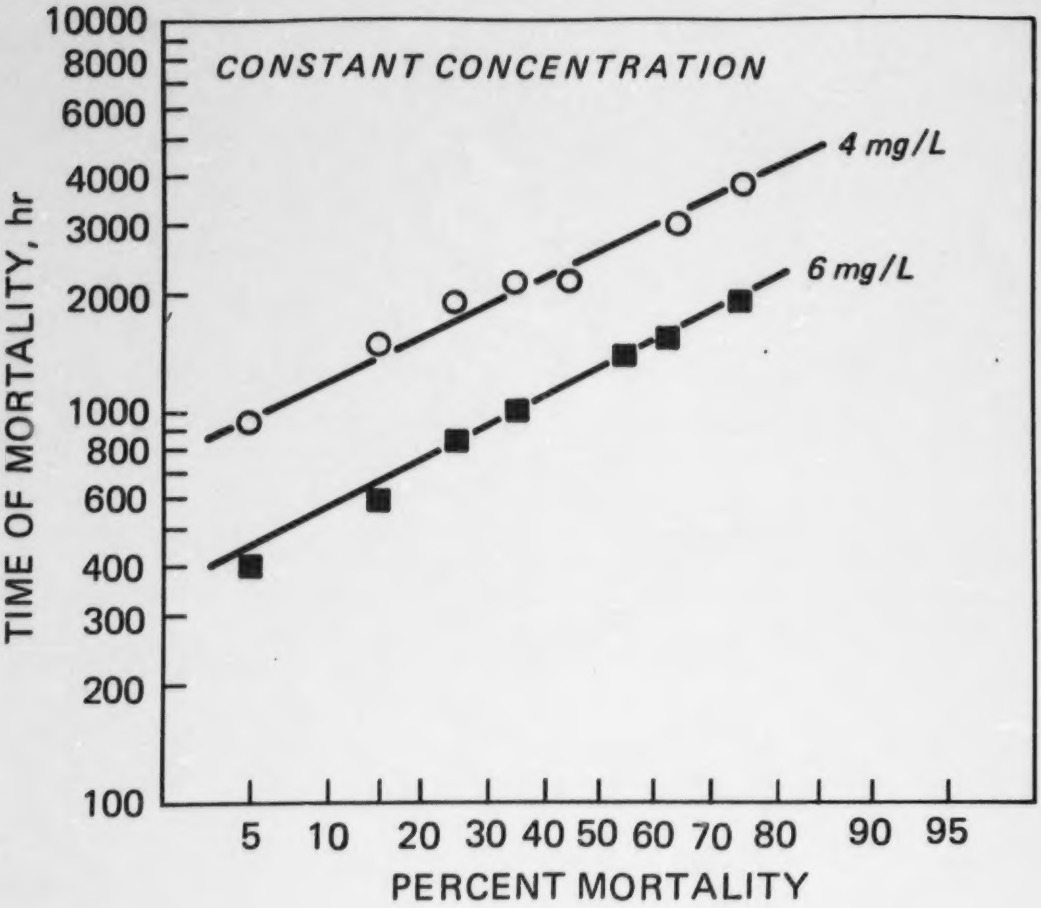


Figure 18. Trout Mortality for Exposures to Constant Zinc Concentrations

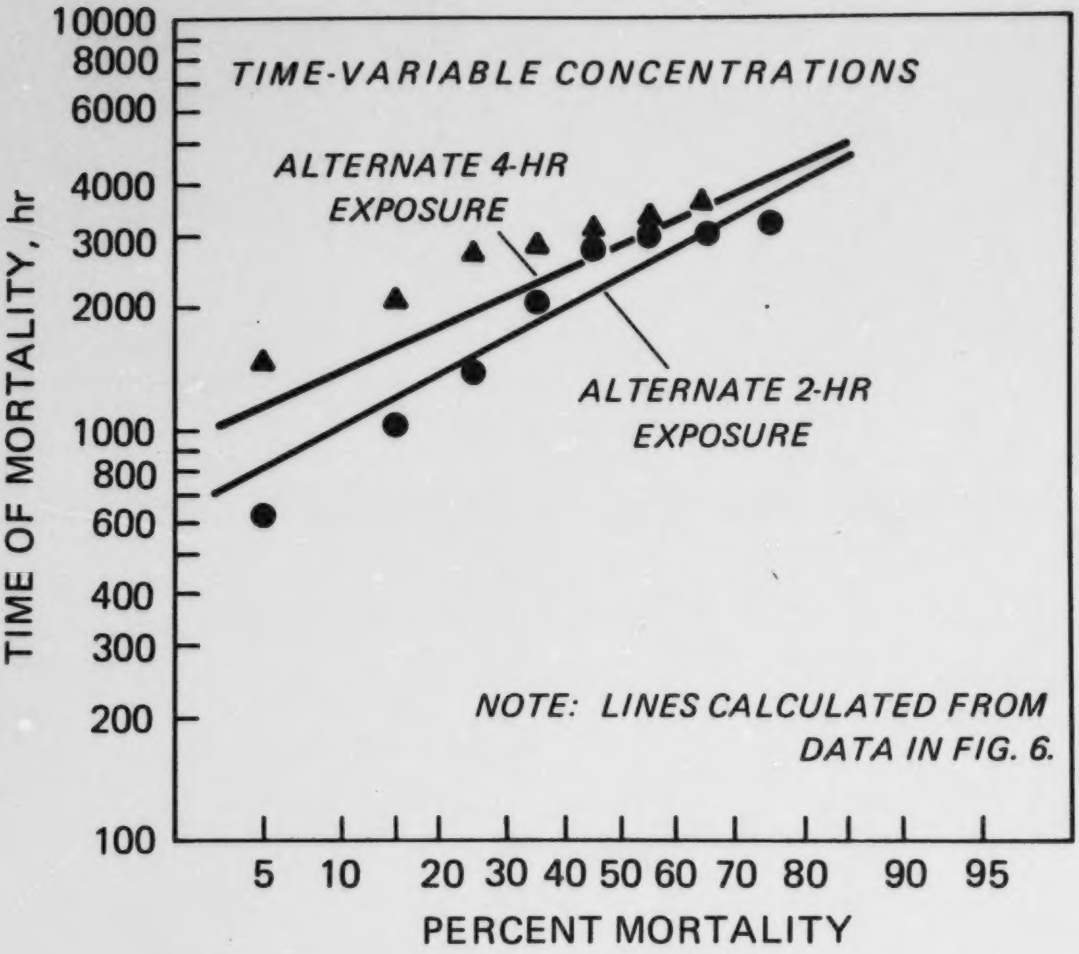


Figure 19. Trout Mortality for Exposure to Time Varying Zinc Concentrations

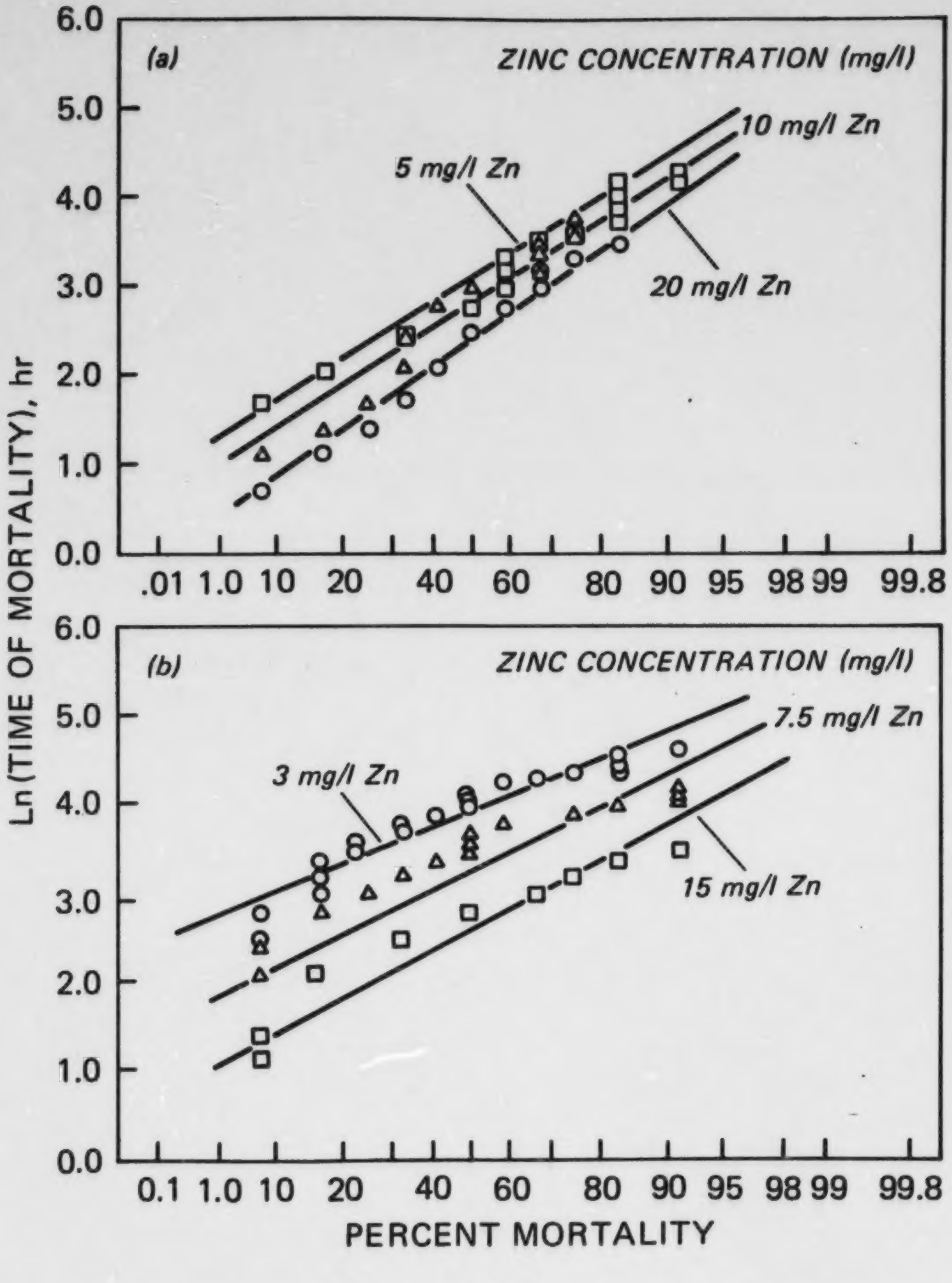


Figure 20. Fish Mortality for Exposures to Constant Zinc Concentrations

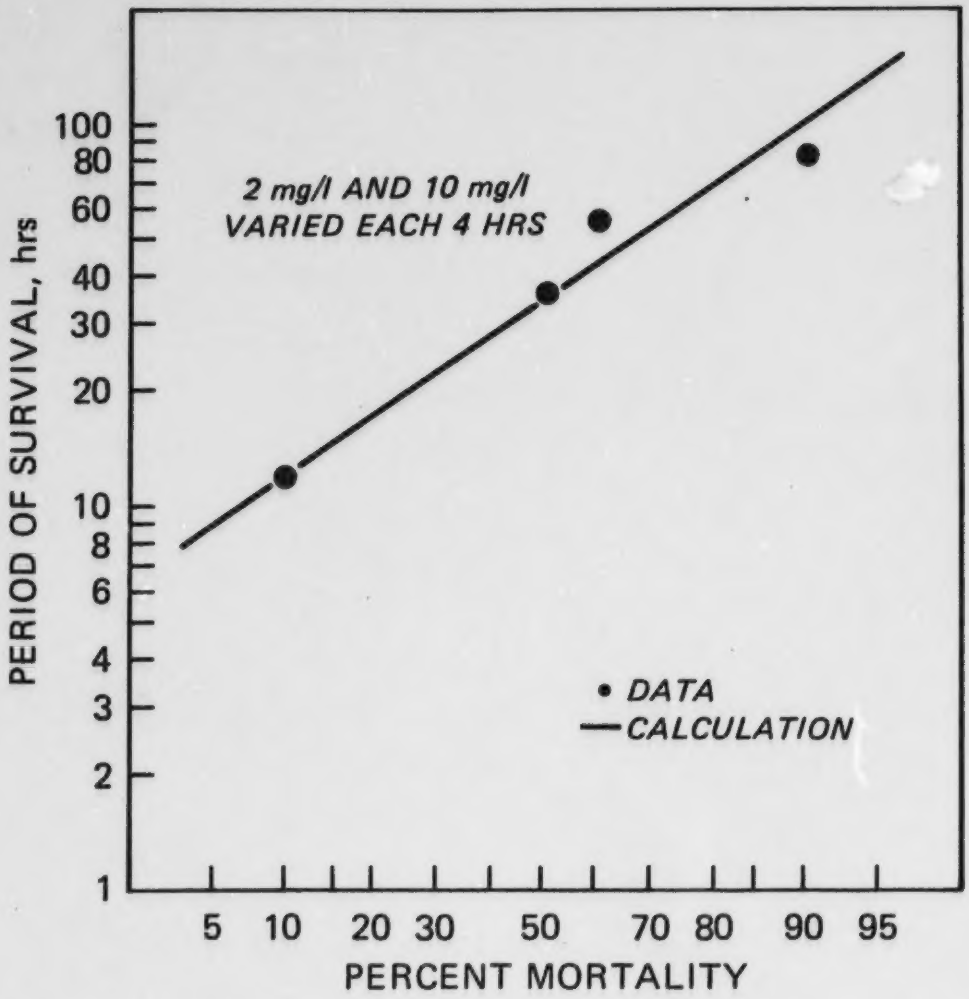


Figure 21. Fish Mortality for-Exposures to Time Varying Zinc Concentrations at 2 mg/ℓ and 10 mg/ℓ

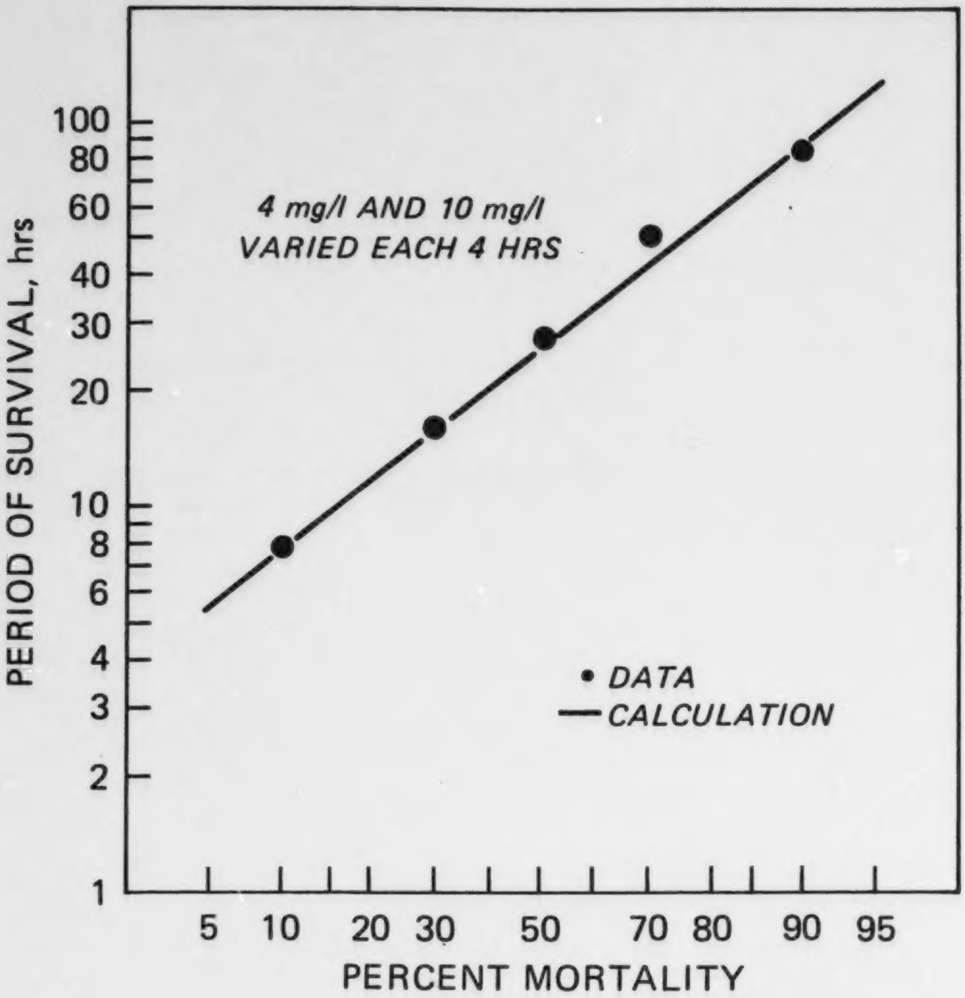
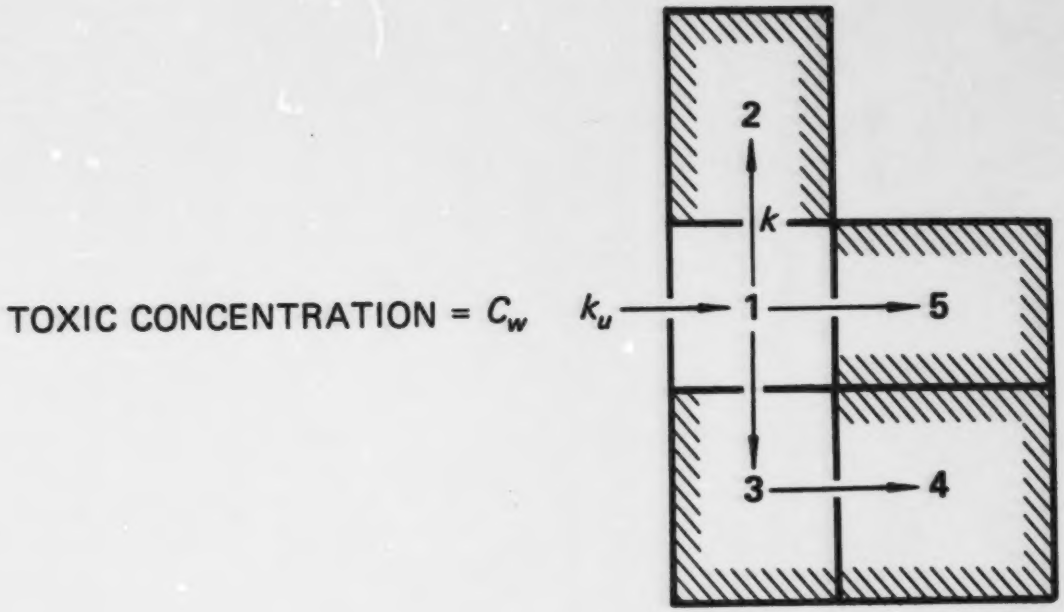


Figure 22. Fish Mortality for Exposures to Time Varying Zinc Concentrations at 4 mg/ℓ and 10 mg/ℓ



Simulation No. 1: (a) All transfers 1st order.
(b) Specified level in Segment 4 yields "effective death."

Simulation No. 2: (a) All transfers 1st order except Segment 3 to Segment 4. There Michaelis-Menton transfer kinetics are used.
(b) Specified level in Segment 4 yields "effective death."

Figure 23. Compartmented Organism Representation

and (36) for "$c_w^{1/2}$" and equations (37) and (38) for "$c_w^{1/4}$". These equations can be derived from integration of equation (30) after making the appropriate substitutions for c_w.

$c_w^{1/2}$ replaced c_w

$$\frac{c_{Nt}}{k_u} = \frac{c_w^{1/2}}{k_R} [1-e^{-k_R t}] \tag{35}$$

and

$$\frac{c_{Nt2}}{k_u} = \frac{c_w^{1/2}}{k_R} [1-e^{-k_R t_1}] + \frac{c_{Nt}}{k_u} e^{-k_R t_o} \tag{36}$$

$c_w^{1/4}$ replaces c_w

$$\frac{c_{Nt}}{k_u} = \frac{c_w^{1/4}}{k_R} [1-e^{-k_R t}] \tag{37}$$

and

$$\frac{c_{Nt2}}{k_u} = \frac{c_w^{1/4}}{k_R} [1-e^{k_R t}] + \frac{c_{Nt}}{k_u} e^{-k_R t_o} \tag{38}$$

The reason some of the data requires kinetic approximations of the uptake from the water phase with c_w to a power "n" (such as c_w^n where n = 1/2 or n = 1/4 or other values of "n") is illustrated by figure 23. This system represents uptake of a toxic from water with the organism divided into multiple compartments represented by segments 1 to 5. When the level of toxic in segment 4 reached a predetermined specified concentration, "effective death" of the simulated organism was assumed. Computations with various concentrations "c_w" outside the organism were developed to determine the time of "effective death" using first order transfers between all organism segments (as shown in the figure). The mortality data generated by this simulation could be represented employing $c_w^{1/2}$ which is equivalent to using equation (35). The calculated mortality could not be represented employing "c_w" which is equivalent to using equation (33). An additional simulation was developed with the compartmentalized organism illustrated in figure 23. The only change was that the transfer of toxic between organism segments 3 and 4 was governed by Michaelis-Menton kinetics. The calculated time and effective death results could be represented employing $c_w^{1/4}$ but not by employ- $c_w^{1/2}$ or c_w.

Therefore, there are feasible internal organism transfer mechanisms which could appear to yield simplified mechanism approximations requiring c_w^n.

A final formulation which may be of interest is associated with a toxicant uptake by an organism with no detoxification. In this instance the equation representing the constant exposure data takes the form of equation (39).

$$\frac{dc_N}{dt} = k_u c_w \tag{39}$$

integration between the limits

$$t = o \quad c_N = o$$

$$t = t_o \quad c_N = c_D$$

yields:

$$t_D = \left(\frac{c_D}{k_u}\right) \frac{1}{c_w} \tag{40}$$

No lethal bioassay data has been encountered in this study which suggests that equation (40) is an appropriate approximate mechanism to represent mortality. It is anticipated that this formulation may be found useful as more data are analyzed.

In summary, there is a family of functions, represented by equations (33) to (40) and perhaps others, which provide reasonable mechanistic approximations for the complex processes that result in organism mortality. These approximations fit data from lethal bioassay tests conducted with exposure to constant concentrations of toxics. Identification of an appropriate mechanistic approximation and evaluation of the coefficients can provide a framework for calculation of probable mortality from exposure to time variable concentrations. Available data on mortality for both constant and time variable exposures to toxics have been employed to test the framework. Additional experiments could provide further information to test the adequacy of the proposed framework.

TABLE 6

Analysis of Some Bioassay Data Using the Proposed Framework

VALUES FOR 50% SENSITIVITY

Ref.	Chemical	Organism	$\frac{1}{Cw^n}$	Cd/Ku	Kr	Hardness
			n			
7	$CuSO_4 \cdot 5H_2O$	bluegill	1.0	80	0.127	360
7	$CuSO_4 \cdot 5H_2O$	fathead minnows	1.0	0.659	0.0314	20
7	$CuSO_4 \cdot 5H_2O$	fathead minnows	1.0	31.676	0.044	360
7	Copper Cyanide	fathead minnows	1.0	25.062	0.0531	14
7	$Cu(NO_3)2$	banded killifish	1.0	22.966	0.0359	53
7	$Cu(NO_3)2$	striped bass	1.0	150.15	0.0281	53
7	Copper sulfate	striped bass	1.0	31.898	0.0147	35
7	$Cu(NO_3)2$	pumpkinseed	1.0	50.76	0.0522	53
7	$Cu(NO_3)2$	white perch	1.0	200	0.0299	53
7	$Cu(NO_3)2$	carp	1.0	39.84	0.0151	53
7	$Zn(C_2H_3O_2)2$	fathead minnows	1.0	14.895	0.0494	20
7	$ZnSO_4 \cdot 7H_2O$	fathead minnows	1.0	13.053	0.0543	20
7	Zinc	cutthroat trout	1.0	9.604	0.0176	
7	$PbC\ell_2$	bluegill	1.0	7171.9	0.0516	360
7	$PbC\ell_2$	bluegill	1.0	377.39	0.0523	20
7	$PbC\ell_2$	guppies	1.0	401.841	0.04524	20
7	$PbC\ell_2$	goldfish	1.0	681.13	0.0399	20
7	Copper sulfate	mosquitofish	1.0	2004.86	0.0332	
7	$CuSO_4 \cdot 5H_2O$	guppies	0.50	6.992	0.02422	20
7	$CuSO_4 \cdot 5H_2O$	goldfish	0.50	4.957	0.0338	20
7	$ZnSO_4 \cdot 7H_2O$	fathead minnows	1.0	487.3	0.0555	360
7	$ZnSO_4 \cdot H_2O$	fathead minnow eggs	1.0	76.05	0.02	174-198
7	$ZnSO_4 \cdot H_2O$	fathead minnow fry	1.0	14.46	0.0509	174-198
7	Zinc	fathead minnows	1.0	131.1	0.0488	
7	$ZnSO_4 \cdot 7H_2O$	bluegills	1.0	95.18	0.0477	20
7	$ZnSO_4 \cdot 7H_2O$	bluegills	1.0	574.06	0.0572	360
7	$Zn(NO_3)2$	killifish	1.0	348.58	0.047	53
7	$Zn(NO_3)2$	striped bass	1.0	221.6	0.0283	53
7	$Zn(NO_3)2$	pumpkinseed	1.0	390.87	0.0445	53

TABLE 6 (cont'd)

VALUES FOR 50% SENSITIVITY

Ref.	Chemical	Organism	$\frac{1}{C_w{}^n}$	Cd/Ku	Kr	Hardness
			n			
11	Zn	guppy	1	39.46	0.00713	
11	CN	guppy	1	4.612	0.04595	
18	Pb	fish @ 27°	1	265.3	0.00135	
17	Sodium-penta chlorophenate	minnows	1	2.73	0.088	156
17	Sodium-penta chlorophenate	goldfish	1	4.87	0.044	148
17	Sodium-penta chlorophenate	bluegill	1	3.80	0.06	145
17	Sodium cyanide	minnow	1	0.3567	0.3512	154
17	Sodium cyanide	catfish	1	1.02	0.156	148
17	Sodium cyanide	goldfish	1	3.91	0.075	152
17	Selenium dioxide	trout	1	467.3	0.0279	148
17	Selenium dioxide	goldfish	1	1748	0.0084	148
17	Selenium dioxide	bluegill	1	996	0.027	150
17	Sodium arsenite	goldfish	1	1765	0.0161	148
17	Sodium arsenite	catfish	1	2436	0.0061	140
17	Sodium arsenite	minnows	1	844	0.0195	149
17	Sodium arsenite	trout	1	1046	0.0158	152
18	Copper	amphipods @ 27°	1	2.9	0.016	
18	Ammonia	daphnia @ 5°C	1	19.31	0.085	
17	Berrylium sulfate	minnow	1	1751	0.0145	140
17	Sodium arsenite	bluegill	1	5003	0.0083	147
18	Zn	amphipods @ 27°C	0.25	18.05	0.039	
17	Selenium dioxide	flagfish 27°C	0.25	64.935	0.0266	
18	Zn	daphnia @ 27°	0.5	15.53	0.0125	
19	Zn	trout	1	87.8	0.0354	320
20	Methanol	minnow	1	36496	0.371	
20	Phenol	minnow	1	244	0.1001	

TABLE 6 (cont'd)

VALUES FOR 50% SENSITIVITY

Ref.	Chemical	Organism	$\frac{1}{Cw^n}$ n	Cd/Ku	Kr	Hardness
21	Cadmium	sand shrimp	0.50	28.74	0.0137	
21	Cadmium	starfish	0.50	62.413	0.00476	
21	Cadmium	sandworm	0.50	98.96	0.03165	
21	Cadmium	killifish	0.50	242.095	0.01449	
21	Cadmium	sheepshead minnow	0.50	149.545	0.0407	
21	Cadmium	grass shrimp	0.25	60.253	0.00594	
21	Cadmium	green crab	0.25	61.875	0.01906	
21	Cadmium	oyster drill	0.25	69.822	0.01909	
22	Sodium hydroxide	bluegill (flow thru)	1.0	31.35	1.266	
22	Sulfuric Acid	bluegill (ft)	1.0	13.4	0.8746	
22	Chlorobenzene	bluegill (ft)	1.0	7.503	0.8964	
22	Methylmethacrylate	bluegill (ft)	1.0	254.45	1.2008	
22	Xylene	bluegill (ft)	1.0	22.52	0.6389	
22	Chromic Acid	bluegill (ft)	1.0	48.53	0.9463	
22	Ammonia	bluegill	1.0	32.93	0.1688	
22	Ammonia	bluegill (plug flow)	1.0	52.778	0.1564	
22	Chromic Acid	bluegill (pf)	1.0	278.36	0.3037	
22	Cupric sulfate	bluegill (ft)	1.0	5.6899	0.1159	
22	Cupric sulfate	bluegill (pf)	0.25	2.726	0.531	
22	Sodium hydroxide	bluegill (pf)	1.0	130.364	0.537	
22	Acrylonitrile	bluegill (ft)	0.50	40.198	0.0995	
22	2-Ethoxy-ethyl-acetate	bluegill (ft)	1.0	213.265	0.20098	
22	Xylene	bluegill (pf)	1.0	55.3097	0.4527	
22	2-Ethoxy-ethyl-acetate	bluegill (pf)	1.0	846.02	0.219	
22	Chlorobenzene	bluegill (pf)	1.0	99.404	0.3796	
22	VDC	fish (static)	1.0	1206.44	0.14001	
7	$CuSO_4 \cdot 5H_2O$	bluegill	1.0	13.387	0.048	20

TABLE 6 (cont'd)

VALUES FOR 50% SENSITIVITY

Ref.	Chemical	Organism	$\frac{1}{Cw^n}$ n	Cd/Ku	Kr	Hardness
7	$Zn(NO_3)_2$	white perch	1.0	144.51	0.0689	53
7	$Zn(NO_3)_2$	carp	1.0	248.09	0.0277	53
7	$ZnSO_4 \cdot 7H_2O$	goldfish	1.0	135.5	0.0409	20
7	$ZnSO_4 \cdot 7H_2O$	guppy	1.0	59.69	0.0173	20
7	Zinc Sulfate	southern platyfish	1.0	460.79	0.023	166
7	Zinc Chloride	bluegills	1.0	112.56	0.0416	20
7	NaCN	trout	1.0	0.99	1.55	
17	SeO_2	minnows	1.0	589.3	0.0008	151
18	Zinc	amphipod @ 27°	1.0	37.92	0.0003	

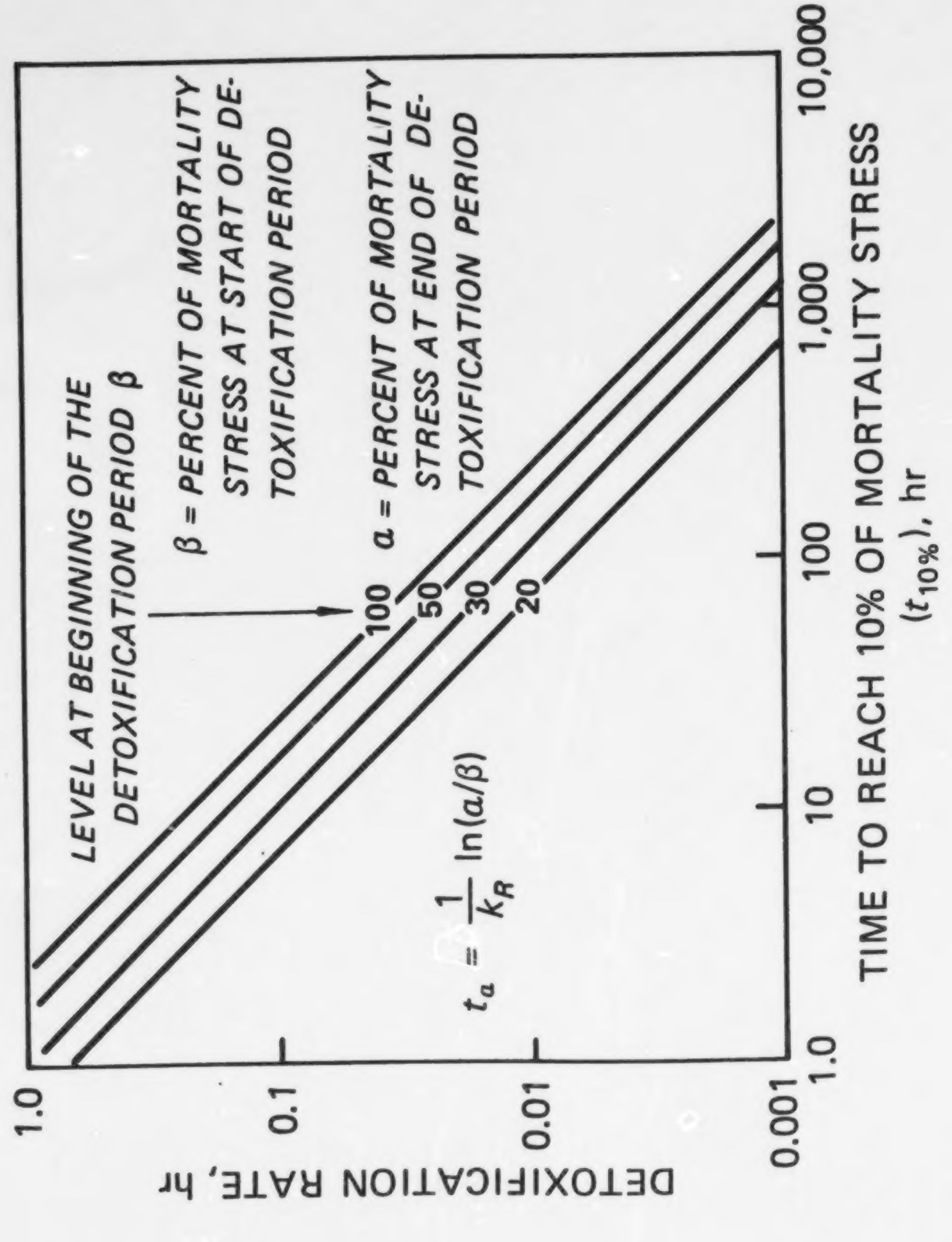


Figure 24. Detoxification Rate vs. Time Required for 90% Reduction of the Effects of the Initial Exposure

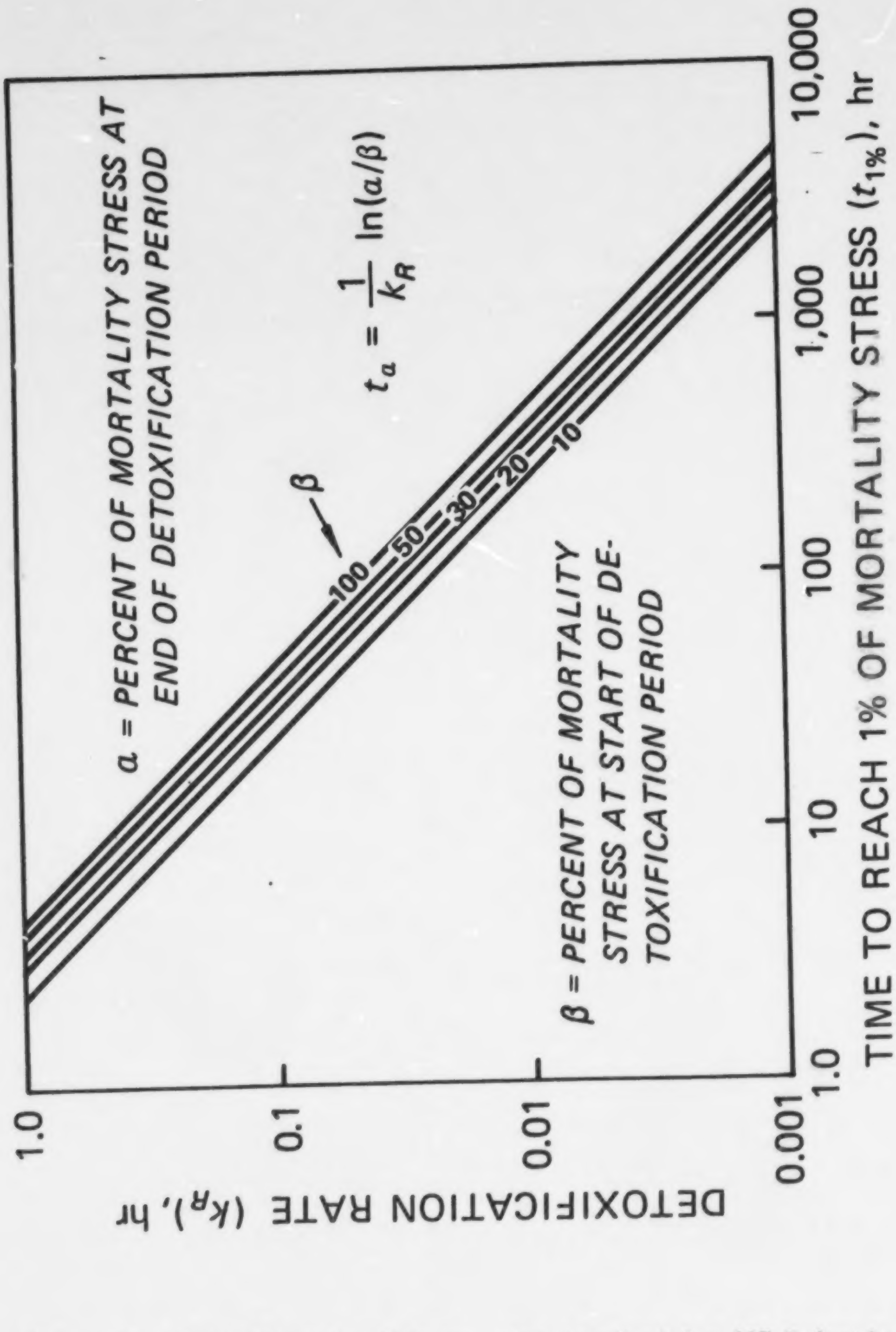


Figure 25. Detoxification Rate vs. Time Required for 99% Reduction of the Effects of the Initial Exposure

Examination of Bioassay Data:

The analysis framework has been employed to examine a small portion of the available data base for lethal bioassay results. The results of this analysis are tabulated in table 6, for the fifty percent sensitivity level of the population. The results are for a heterogeneous data base. In some instances the detailed results of the mortality experiments were available for analysis. In other instances, the only results available were the author's estimates of LC_{50} for three periods of exposure.

The detoxification rate "k_R" varies from .001/hr to 1.6/hr. This rate "k_R" controls the time required for the organisms to reduce the internal level of the toxic. Consider a series of exposures and the question: How long must the period between exposures be so that the effect of the previous exposure is reduced to the point (say 10% of the "equivalent mortality dose") so that each exposure can be considered as independent? Figures 24 and 25 present the calculated time between exposures required to reduce the effects of the initial exposure to 10% and 1% of the "equivalent mortality dose". For k_R between 1.0 and 0.1 the times required are on the order of 10 and 40 hours for the 10 and 1% levels respectively. By contrast for "k_R" between 0.01 and 0.001 the times required are on the order of 1000 and 3000 hours for the 10 and 1% levels respectively. From a practical standpoint, for most substances, it appears that long times between exposures are needed to make two exposures independent. This suggests that for problems like urban runoff, CSO discharges, frequent spills or continuous discharges, the impact of a series of events are not independent. By contrast, for a few substances and organisms with large detoxification rates, "k_R" exposures separated by as little as 10 hours may be viewed as independent events.

Applications and Limitations:

The proposed framework can be employed to calculate the probable mortality, and/or the percentage of mortality stress anticipated from exposure to any time history of toxic concentrations. Problems which can be viewed in the context of the proposed framework include:

1. Organism responses (as a percentage of the mortality level) for continuous discharges when both load and flow vary as a function of time. (Food chain effects have not been included.)
2. Analysis of the effects of toxic spills.
3. Organism responses to CSO and urban runoff discharges.
4. Determinations of mixing zone sizes considering fish swim through time.
5. The concentration required for an effect (mortality or other effect) can be calculated for various exposure periods. The effects from previous exposures can be considered if required.

The proposed framework has been tested against two independent data sets. Additional data sets which have information on exposure to both constant and time variable toxic concentration levels should be developed to test the proposed framework. Many of the problems inherent in relating bioassay data obtained in the laboratory to field conditions are not altered or solved by the proposed framework. This includes chemical speciation, identification of sensitive biological species, acclimation, synogistic, or antagonistic effects. The method proposed does allow utilization of the vast existing bioassay data base, both chronic and acute, to predict effects associated with time varying exposure concentrations. The proposed framework can also accommodate bioassay data developed in terms of % wastewater or toxic units.

SECTION 7

RELATIONSHIP BETWEEN FISH MORTALITY AND LOW DISSOLVED OXYGEN LEVELS

Experimental investigations which explore the relationship between low dissolved oxygen levels and fish mortality are similar, in concept, to bioassays of toxic substances. Individuals from a population of test fish are randomly distributed between a number of experimental chambers. The level of dissolved oxygen is controlled in each chamber and represents the environmental stress to which the fish are subjected. Mortality is observed as a function of time in each chamber. Usually, mortality occurs at different times for the individual fish in each experimental chamber, as illustrated by the data in figure 26. The concept of a population sensitivity distribution can also be applied to fish mortality resulting from exposure to low dissolved oxygen levels. Using similar reasoning to that developed for exposure to toxics, the characteristics of individuals within the population should be similar for those at the same sensitivity level and should differ between individuals at different sensitivity levels.

The detailed mechanisms involved in the transfer of oxygen from the water through the gills, to the blood and ultimately to sites in the fish body are complex (23,24). The approach taken in this study is to attempt to identify relatively simple mechanistic approximations which can be employed to represent the processes involved.

A representation which appears to be consistent with the observed data considers buildup of a respiratory requirement or toxic by-product "R_N" in fish. It is assumed that this requirement can be defined by:

$$\frac{dR_N}{dt} = k - k_R R_N \tag{39}$$

$$\text{where } k = \frac{k_T}{C_w} \tag{40}$$

$$\therefore$$

$$\frac{dR_N}{dt} = \frac{k_T}{C_w} - k_R R_N \tag{41}$$

where:

R_N = toxic by-product $\frac{\text{Mass}}{\text{Fish Mass}}$

k_R = detoxification rate (1/Time)

k = toxic by-product production $\left(\frac{\text{Mass}}{\text{Fish mass-time}}\right)$

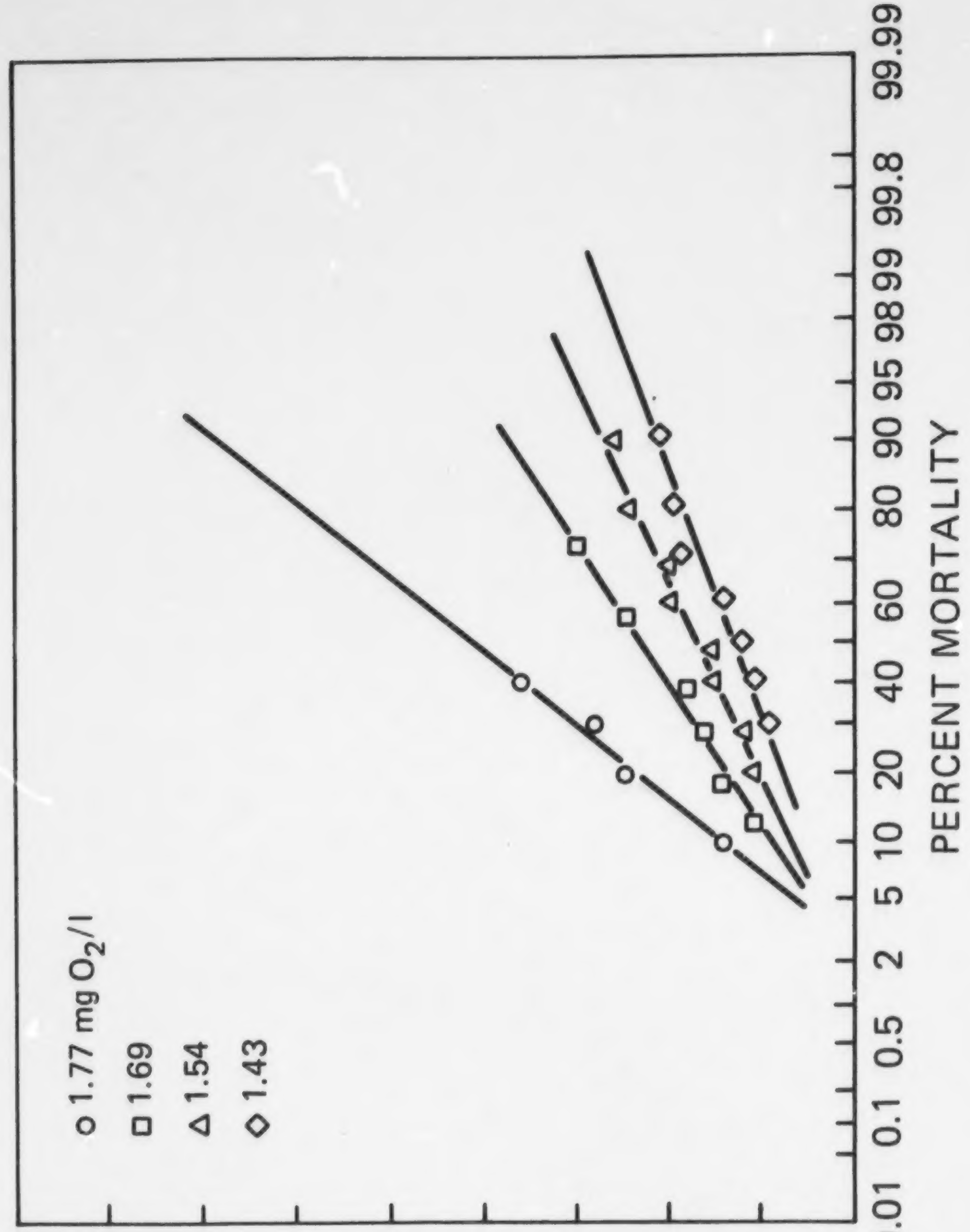


Figure 26. Fish Fry Mortality for Exposure to Constant Oxygen Concentrations

Equation (40) provides an increase in the rate of toxic by-product production under conditions of stress caused by low dissolved oxygen. For example, this could be associated with increased water pumping across gill surfaces at low dissolved oxygen levels. For equation (40):

k_T = base toxic by-product production rate $\left(\frac{\text{mass-vol}}{\text{Fish mass-}O_2\text{mass-time}}\right)$

C_w = dissolved oxygen concentration $\left(\frac{O_2\text{mass}}{\text{vol}}\right)$

Integration of equation (41) between the limits:

$t = o \qquad R_N = 0$

$t = t_o \qquad R_N = R_D$

and solving for Cw in terms of t_D yields:

$$C_w = \frac{k_T}{R_D} \cdot \frac{1}{k_R} (1 - e^{-k_R t_D}) \tag{42}$$

This equation is similar to that employed for analysis of the effects of toxics substances and can be obtained by substituting "1/Cw" for "Cw" in equations (30) to (34). The unknown parameters "k_R" and "R_D/k_T", which are obtained from the observed fish mortality data have meanings which are comparable to those discussed under the toxics analysis. "R_D/k_T", is an "equivalent mortality dose". Figure (27) illustrates the comparisons which can be obtained between calculated and observed results on time of fish mortality and dissolved oxygen concentration.

Sequences of exposure to time variable dissolved oxygen concentrations can be analyzed employing equations (43) and (44):

$$\frac{R_N}{k_T} = \frac{1}{C_w \cdot k_R} (1 - e^{-k_R t}) \tag{43}$$

$$\frac{R_{N}t_1}{k_T} = \frac{1}{C_{w1} \cdot k_R} (1 - e^{-k_R t_1}) + \frac{R_N}{k_T} e^{-k_R t_1} \tag{44}$$

Equation (43) is employed for the initial calculation of "$\frac{R_N}{k_T}$". This

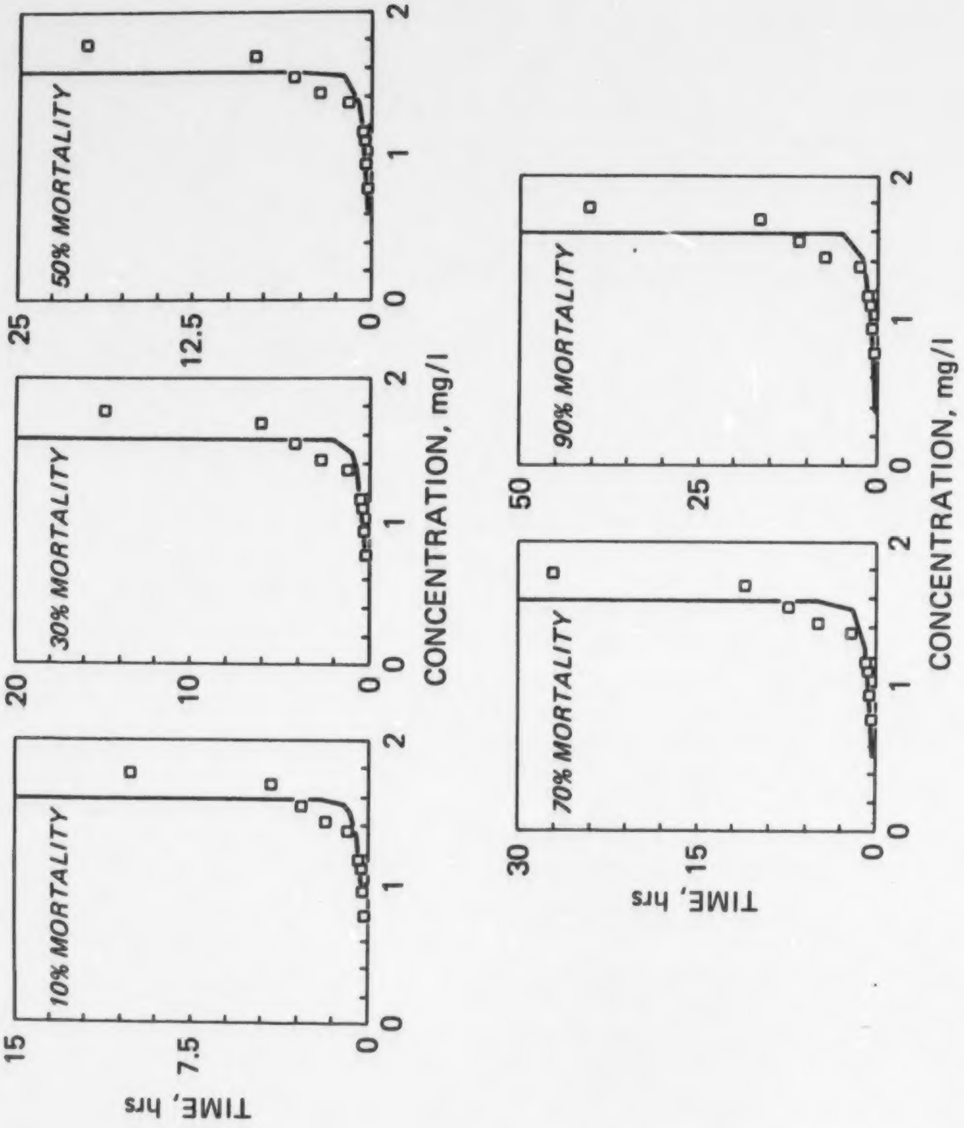


Figure 27. Dissolved Oxygen Toxicity of Speckled Trout

value is substituted into equation (44) for the next exposure during time "t_1". $\frac{R_N t_1}{k_T}$ is calculated and can be used to replace "$\frac{R_N}{k_T}$" for the following exposure. The procedure can be repeated for any sequence of dissolved oxygen concentrations.

The analysis framework relating dissolved oxygen and fish mortality does not have the same level of physical significance as the comparable frameworks presented for the analysis of the effects of toxic substances. In addition the dissolved oxygen framework has not been tested employing time variable concentration data. This framework therefore should be considered provisional. It has however, satisfactorily analyzed a representative number of data sets relating fish mortality to dissolved oxygen as indicated in the following section.

Analysis of Dissolved Oxygen Bioassay Data

The dissolved oxygen framework has been utilized in the analysis of several experimental data sets relating fish mortality to low dissolved oxygen (25,26,27). The type of data available differed slightly in each case. Shepard (25) reported per cent mortality vs. time for various oxygen levels. An example of this data is shown in fig. 26. Downing and Merkins (26) examined the influence of temperature on the survival of several species of fish in low dissolved oxygen and reported the median period of survival vs. oxygen concentration for the 50% mortality level. The data presented by Moss and Scott (27) are results of shock tests conducted to determine the critical oxygen level (the lowest D.O. level at which fish consistently survived for 24 hours). Therefore, the plots of survival time vs. dissolved oxygen actually represents the conditions under which there was no survival.

Even though the data sets analyzed were not homogeneous, they can all be represented by the mechanistic framework outlined in the previous section. The results are tabulated in Table 7. Figures 27 and 28 illustrate the comparison obtained between the experimental data and calculated function using the constant values of the two coefficients shown on the figures.

The detoxification rate, "k_r" varies from 0.006/hr. to 3.7/hr., approximately the same as the range of values for the toxic substances analysis. This rate controls the time between events which is required before the effects of events can be considered as separate. The detoxification rate is between 0.1 and 1.0 for half of the data sets analyzed. Based on figures 24 and 25 this requires a time of 2 - 40 hours for the effect of one exposure to have no influence on the following event. Fifteen percent of the cases examined require 40 - 200 hours between exposures of low dissolved oxygen for them to be considered as independent events ($k_r < 0.01$). In general, the time between exposures required to have independent events from an effects standpoint, appears to be shorter for dissolved oxygen than for toxics.

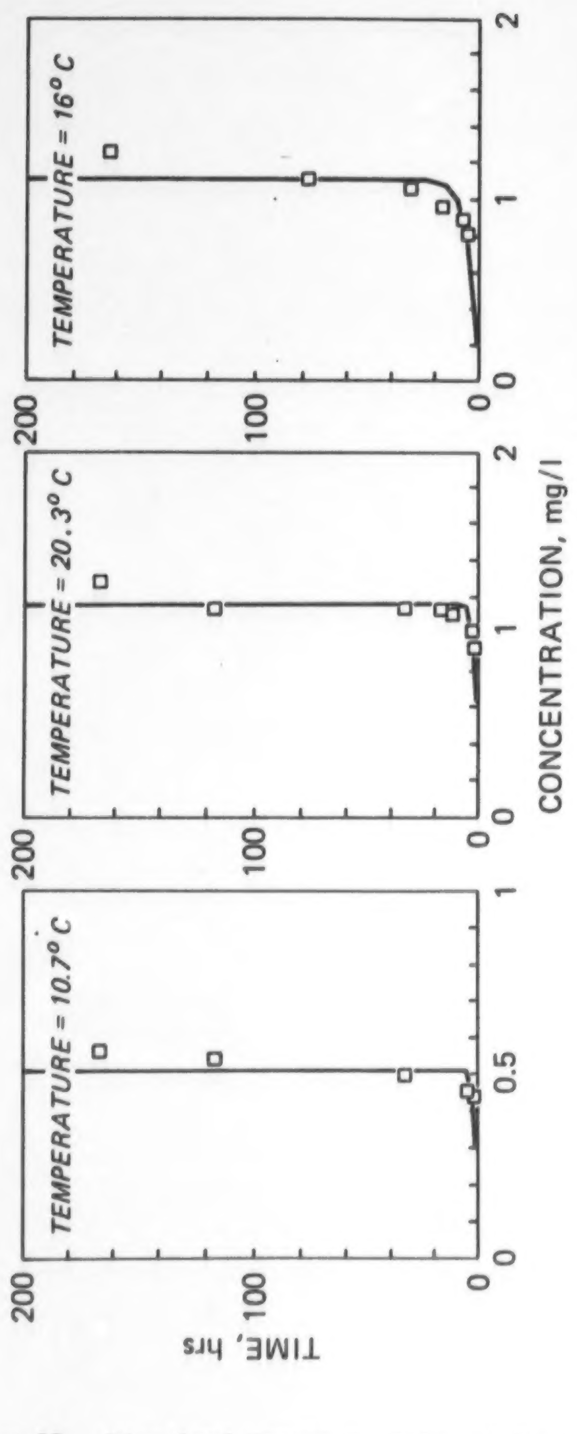


Figure 28. Dissolved Oxygen Toxicity of Perch

TABLE 7

Summary of Results of D.O. Toxicity Data Analysis

Reference	Organism	% kill	R_D/k_u	k_R
Shepard	Speckled trout	10%	0.171	3.69
		30%	0.234	2.71
		50%	0.315	2.01
		70%	0.394	1.60
		90%	0.59	1.06
Moss & Scott	Bluegills @ 25°	100%	2.3776	0.6012
	Bluegills @ 30°C	100%	0.8634	1.3662
	Largemouth Bass @ 25°C	100%	2.6021	0.4416
	Largemoutn Bass @ 30°C	100%	0.5653	1.5554
	Largemouth Bass @ 35°C	100%	0.4043	1.7587
	Channel Catfish @ 25°C	100%	1.6223	0.6866
	Channel Catfish @ 30°C	100%	1.7457	0.5887
	Channel Catfish @ 35°C	100%	0.3709	2.5709
Downing & Merkens	Mirror Carp @ 10.6°C	50%	5.43	0.437
	Mirror Carp @ 16°C	50%	41.66	0.019
	Mirror Carp @ 19.9°C	50%	33.33	0.0061
	Chub Fish @ 20.5°C	50%	6.33	0.1418
	Dace fish @ 16.2°C	50%	0.929	1.4695
	Perch @ 10.7°C	50%	2.137	0.9276
	Perch @ 16°C	50%	4.095	0.2218
	Perch @ 20.3°C	50%	1.0454	0.8278
	Roach fish @ 10.7°C	50%	21.978	0.0938
	Roach fish @ 16.1°C	50%	2.7427	0.5006
	Roach fish @ 19.1°C	50%	2.6295	0.2725
	Tench fish @ 9.4°C	50%	100.55	0.044
	Tench fish @ 16°C	50%	7.2939	0.3033
	Rainbow Trout @ 10.6°C	50%	0.8486	0.7557

Computations for exposures to time variable oxygen concentrations can be undertaken employing equation 44, assuming a specified k_r and initial equivalent dose "R_N/k_T" (which can be zero). Figure 29 shows results of the computation simulating two 6-hour exposures at a low oxygen level of 0.5 mg/ℓ with 60 hours between each exposure at 5.5 mg O_2/ℓ for two depuration rates. Note that in both cases 60 hours is sufficient time between exposures so each event is independent. As expected, the equivalent dosage computed at both oxygen concentrations are larger for k_r = 0.3/hr than for the case of k_r = 2.0. Based on the summary of data in Table 7, a 6-hour exposure to 0.5 mg/ℓ for a population whose depuration rate is 0.3/hr. will kill 77% of the organisms. At k_r = 2.0/hr only 41% of the population is killed. These kills will take place during the first storm event.

As previously noted, the physical interpretation of the analysis framework for fish mortality as a result of exposure to low dissolved oxygen levels is weak. The framework has satisfactorily analyzed a representative set of constant concentration data but time variable applications have not yet been tested.

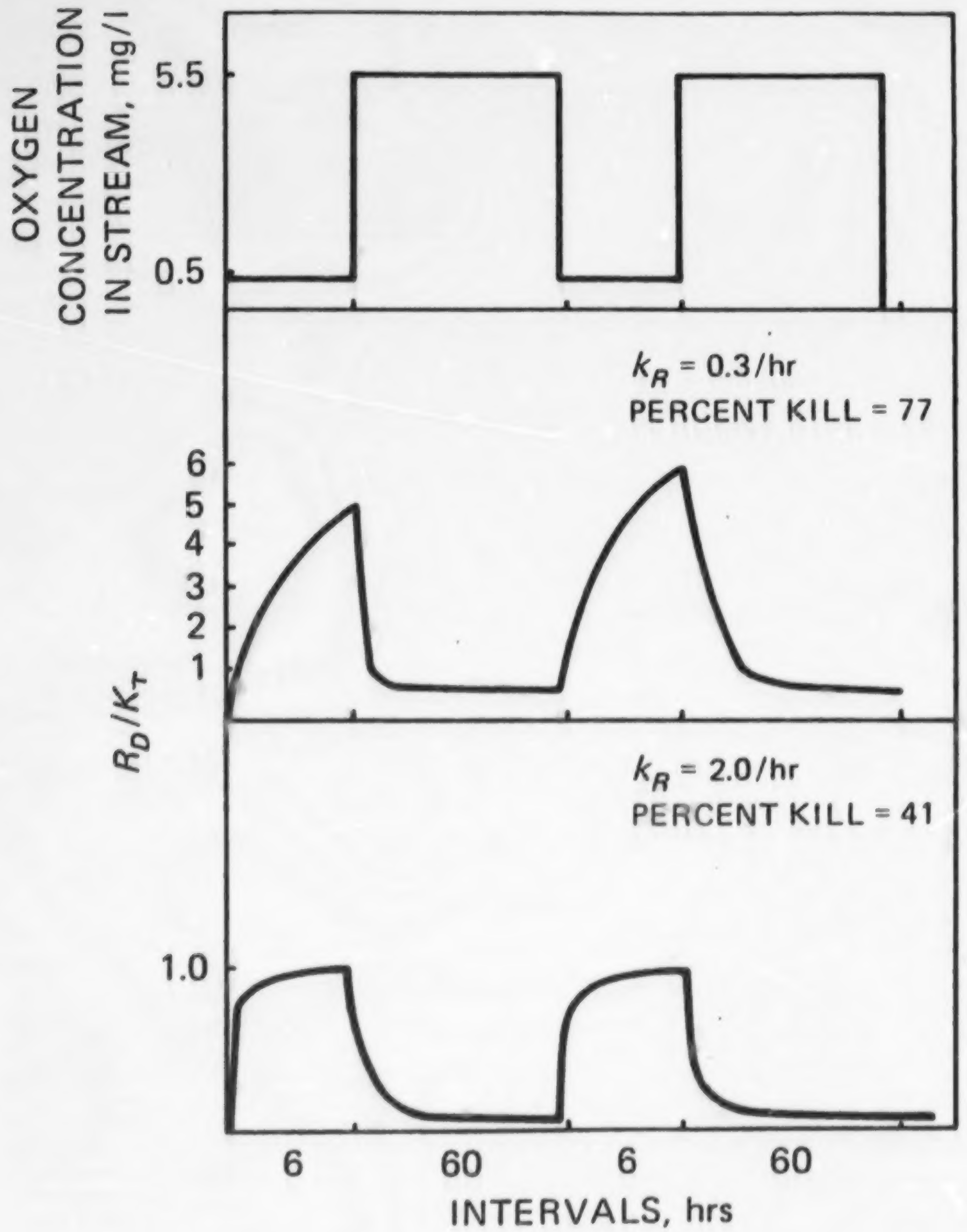


Figure 29. Effects of Exposure to Time Varying Oxygen Concentrations on Fish Mortality

SECTION 8

WATER QUALITY CRITERIA FOR INTERMITTENT URBAN DISCHARGES

Statement of Issues

Urban discharges associated with rainfall are characterized by relatively short duration events with relatively large time periods between events. Table 8 has been prepared to illustrate some of the times involved:

TABLE 8

URBAN DISCHARGE TIME FACTORS

Approximate National Average Storm Duration	4.5 hours
Average Storm Duration in Some Regions	6 hours
∿ 90% Storm Duration	12 to 12.5 hours
∿ Average Time Between Storms	60 hours

The time factors which characterize storm related urban discharges are very different from those probably considered in developing generalized water quality criteria and standards. Therefore, a question exists as to: what are appropriate water quality criteria for a highly time variable discharge such as storm related urban discharges? The significance of this question is reinforced by the potential cost of treatment and by the alternative definitions of water quality problems which can be considered. These alternative problem definitions are:

1. Water quality which results in impairment or loss of beneficial usages.
2. Water quality which results in violations of numerical standards.

In the context of the above discussion, several levels of water quality criteria could be considered. The criteria include those promulgated by EPA as generalized criteria, and, in addition, criteria which reflect the time factors that characterize storm related discharges and are related to impairment of beneficial usage.

The discussions that follow provide information on each of the water quality criteria sets which could be considered. There are criteria for recreation which have not been considered since they are normally for coliform organism densities. The concentration of coliform in both CSO and Urban runoff are usually orders of magnitude above criteria thus site specific analysis would usually be most appropriate.

Generalized Water Quality Criteria

Table 9 summarizes information on numerical criteria promulgated by EPA in the "Red Book" and in the Federal Register for toxic constituents. The information is limited to major contaminants in storm related urban discharges. Also indicated in Table 9 are the mean concentration and 84% concentration in storm related urban discharges as determined from the University of Florida data base. For urban runoff, except for arsenic and chromium; the expected mean concentration is higher than the maximum value allowed by the criteria. If a literal interpretation of the 24 hour concentration is used and the average duration of a storm is 4.5 hours, then the event average concentration would have to be no greater than 5.3 times the 24 hour criteria concentration to yield a 24 hour average concentration equal to the criteria. For zinc, copper, lead, and cadmium, violation of the 24 hour criteria could also be calculated.

TABLE 9. PROMULGATED WATER QUALITY CRITERIA

Chemical	Criteria (mg/ℓ)		Expected Concentration in Urban Runoff (mg/ℓ)		CSO(mg/ℓ)
	Max.	24 hr.	Mean	84%	Mean
Zinc	0.18	.05	.163	.283	.68
Arsenic	.44	.04	.05	.06	-
Copper	.012	.006	.03	.04	.38
Lead	.074	.0008	.325	.526	.56
Chromium	2.2 (TFI) .02(HEX)	.04 (TRI) .0003(HEX)	.018(TOT)	.03	-
Cadmium	.0015	.000012	.003	10046	.03

For CSO the mean concentrations for zinc, copper, lead, and cadmium are above the maximum generalized criteria. Due to the variability of CSO event mean concentrations discussed under the Pollutant Loads section of the report frequent concentration levels in excess of the generalized standards would be anticipated for both the maximum and 24 hour average criteria.

The comparisons presented above consider undiluted storm related urban discharge. The results of an analysis which considers dilution will be presented subsequently.

Impairment of Beneficial Use Criteria

Impairment of beneficial use will, for the following discussion, be considered concentrations that result in mortality of 50% of the population. Other criteria, such as no mortality, could also be developed and employ similar calculation procedures.

The analysis framework presented in the report section on biological responses to time variable concentrations was employed together with the data on equivalent mortality dosage, detoxification rates (from table 6) and expected mean concentrations in urban runoff and CSO. A series of calculations were developed which considered exposure of the more sensitive fish to a series of average duration storm events having the expected mean concentration of each contaminant. The interval between storms was 60 hours. The calculated equivalent dosage was allowed to stabilize and the concentration required to produce mortality at the 50% level of population sensitivity was calculated. The results are summarized in table 10. These results include the effects of carry-over between average storm conditions. The calculated concentrations for mortality are presented for 4.5 and 12 hour storms.

The calculation procedure can be employed to develop criteria for general water bodies or could be applied on a site specific basis considering river or stream dilutions.

TABLE 10. CALCULATED CONCENTRATIONS REQUIRED FOR MORTALITY OF SOME FISH SPECIES AS A RESULT OF EXPOSURE TO URBAN RUNOFF AND CSO DISCHARGES

	Expected Mean(1) mg/ℓ		Concentration (mg/ℓ) for 50% Mortality(2)			
			Urban Runoff Storm		CSO Storm	
Chemical	Urban Runoff	CSO	4.5 HR	12 HR	4.5 HR	12 HR
Zinc	.16	.68	1.8	.8	1.4	.5
Copper	.03	.38	.6	.2	.4	.1
Lead	.33	.56	11.	4.3	7.9	3.
Cadmium	.003	.03	11.3	4.7	11.2	4.5

NOTES: (1) Expected event mean concentration calculations assuming large data base.

(2) Effects of carry-over of expected mean concentrations and other average storm conditions are included.

A comparison of the criteria in table 9 and the calculated concentrations for mortality in table 10 indicate that there are substantial differences ranging from almost an order of magnitude to several orders. This difference is in part a result of the differences in data base used to define sensitive species. Another, equally important, source of this difference is the manner in which the duration of exposure has been included in the analysis.

Neither set of concentrations is satisfactory criteria for storm event related exposures. The generalized criteria do not take into account the time scale of exposures associated with storm events. These criteria tend to be protective of the environment by restricting allowable concentrations during the short exposure periods characteristic of runoff events. Use of the generalized criteria will tend to require higher levels of treatment than may be necessary with associated costs for construction and operation. By contrast, the calculated concentrations presented in table 10 tend to overestimate allowable concentrations since the data base analyzed may not include representative sensitive species which require protection.

Neither the generalized criteria nor the stormwater criteria consider synergistic or antagonistic effects. Perhaps even more significant is the fact that these criteria do not include chemical complexing, absorption etc. which can reduce the biologically effective concentrations of toxicant by large amounts.

Concentration criteria which are appropriate for storm related phenomena would be between the two sets of values. Methods have been developed which could employ the existing data base to calculate criteria which consider time variable concentrations and exposure periods which are consistent with storm event exposure durations and the interval between storms.

Chronic Effects

The usual approach to establishment of water quality criteria considers acute effects such as mortality and chronic effects such as inhibited reproduction, etc. The generalized criteria derive the maximum concentration limit from acute effects protection limits and the 24 hour limit from chronic effects protection limit. The methods developed in this project can be employed to calculate the time history of stress on the organism ("equivalent exposure"). The equivalent mortality dose producing mortality of 50% of the population is obtained from the analysis of bioassay data. The calculated equivalent dose at any time which results from some sequence of exposures can be divided by the equivalent mortality dose. This ratio (as % of equivalent mortality dose) could be considered as a measure of the chronic stress to which the organism is subjected. The interpretation is complicated in that different sites within the organism are usually associated with chronic effects. This is under investigation in an ongoing exploration.

Table 11 presents the calculated % equivalent mortality dose carried over, on average, from a sequence of storms. This is the calculated equivalent mortality dose at the start of a storm event. Table 11 also presents information on the calculated % equivalent mortality dose at the end of 4.5 and 12 hour storms whose concentrations are at the mean expected value. These results suggest that, for some of the toxics analyzed, a variable but moderately high level of stress may result from exposure to the undiluted contaminants in urban runoff. Stresses on the order of 2 to 25% of the equivalent mortality dose could produce some chronic effects. The calculations presented in Table 11 are for undiluted urban runoff. Computations could be developed considering various dilutions of the runoff and CSO.

TABLE 11. CARRYOVER EFFECTS BETWEEN STORMS (URBAN RUNOFF)

Chemical	Expected Mean Concentration (mg/l)	% Mortality Stress Average Carryover	 @ 4.5 hr. Storm	 @ 12 hr. Storm
Zinc	.163	8.6	15.7	26.8
Arsenic	.05	-	-	-
Copper	.03	2.6	6.4	12.2
Lead	.325	8.3	8.8	9.7
Chromium	.018	-	-	-
Cadmium	.003	2.3	3.1	4.4

Dissolved Oxygen Criteria

Water quality criteria for dissolved oxygen fluctuations, such as those that might be associated with urban discharge time factors have not been examined. Usual standards for dissolved oxygen require a minimum of 5 mg/l. Proposed modifications, to D.O. standards, by the state of Ohio (Federal Register Vol. 45, 231, 11/28/80, 79054) for warm water fishery standards on some water bodies of 5 mg/l for 16 hours of any 24 hour period and not less than 4 mg/l at any time were denied by EPA. There is strong historical precedence for maintaining D.O. standards on most water bodies at a minimum of 5 mg/l. This is usually based on information similar to that summarized in Table 12.

An approach to dissolved oxygen water quality criteria similar to that used for toxics can be considered. Generalized criteria for dissolved oxygen would be 5 mg/l. Criteria for storm event time scales should be no less than 2.5 mg/l. The carryover effects of exposure to low dissolved oxygen levels from sequences of storm events appears small as discussed in the report section dealing with the effects of time variable dissolved oxygen on fish.

A major data gap appears to exist with respect to chronic effects associated with the urban discharge time scales of hours (4.5 to 12) of low D.O. and days (60/24 = 2.5) of more normal dissolved oxygen levels.

TABLE 12. SUMMARY OF THE INFORMATION AVAILABLE ON THE EFFECTS OF DISSOLVED OXYGEN CONCENTRATION ON FISH

Dissolved Oxygen	Effects Reported	Reference
Saturation to 5 mg/l	1. Generally considered adequate for a healthy	USEPA Red Book (7)
5 mg/l to 2.5 mg/l	1. Sublethal effects on adults observed in laboratories.	(Abernathy)(28)
	2. Reduced growth rate associated with constant exposure of adults.	(Siefert et al.)(29) (Moss)(26) (Warren)(30)
	3. Some increased mortality of early life stages (no direct data on population effects.	
	4. Time variable exposures (8 to 12 hours every 24 hours) appeared to result in reduced growth rates.	(Whiteworth)(31)
2.5 mg/l to 1.5 mg/l	1. Possible mortality of adult and/or smaller fish due to combination of stresses with significant D.O. contribution to mortality.	(Moss)(26) (Abernathy)(28) (Warren)(30)
1.5 mg/l to zero	1. Fish mortality (short exposure).	(Moss)(26) (Warren)(30)

SECTION 9

REGIONAL ANALYSIS

An analysis of concentrations in rivers and streams has been applied to data from nine regions of the country. Log-normal distributions are employed to represent the rainfall and stream flow for individual regions, contaminant concentrations in the runoff, the resulting dilution, and stream concentration. The available dilution is controlled in part by the ratio of the stream drainage area to the urban drainage area and this concept is included in the calculation.

Monthly statistical characterization of rainfall was available for each region (for example, figures 30, 31, and 32). Estimates of the median and coefficient of variation of rainfall was made from these figures and converted to volume of runoff per unit of drainage area for both CSO and separate sewer systems employing the following calculation.

$$Q_u = \frac{I_v \times A_u \times R_v}{D} \qquad (45)$$

where:

Q_u = Average urban runoff event flow

I_v = Rainfall volume over the event

D = Duration of the runoff event

A_u = Urban drainage area

R_v = Ratio of runoff to rainfall based on impervious area

$R_v = 0.2$ for separate sewer systems

$R_v = 0.4$ for combined sewer overflow systems

This resulted in runoff distributions as defined in table 13, and shown in figures 33, 34, and 35. Statistical data were also available for streams and rivers in each region. The available information included the drainage area for each water body, the mean and median flows, and coefficient of variation for a number of years of records (20-100 years). The rivers were categorized according to the rainfall regions listed in Table 13. The data were converted to cfs/mi^2 and averaged by region generating log probability plots of flow per unit drainage area similar to those for runoff. For each stream/urban area ratio assumed the flow distributions were increased proportionately. The cfs/mi^2 varied within a particular region. From the standpoint of a broad base regional analysis these variations could be ignored. By contrast if a more detailed regional analysis were required ranges of cfs/mi^2 could be used in the analysis to represent the observed variations between streams. Finally

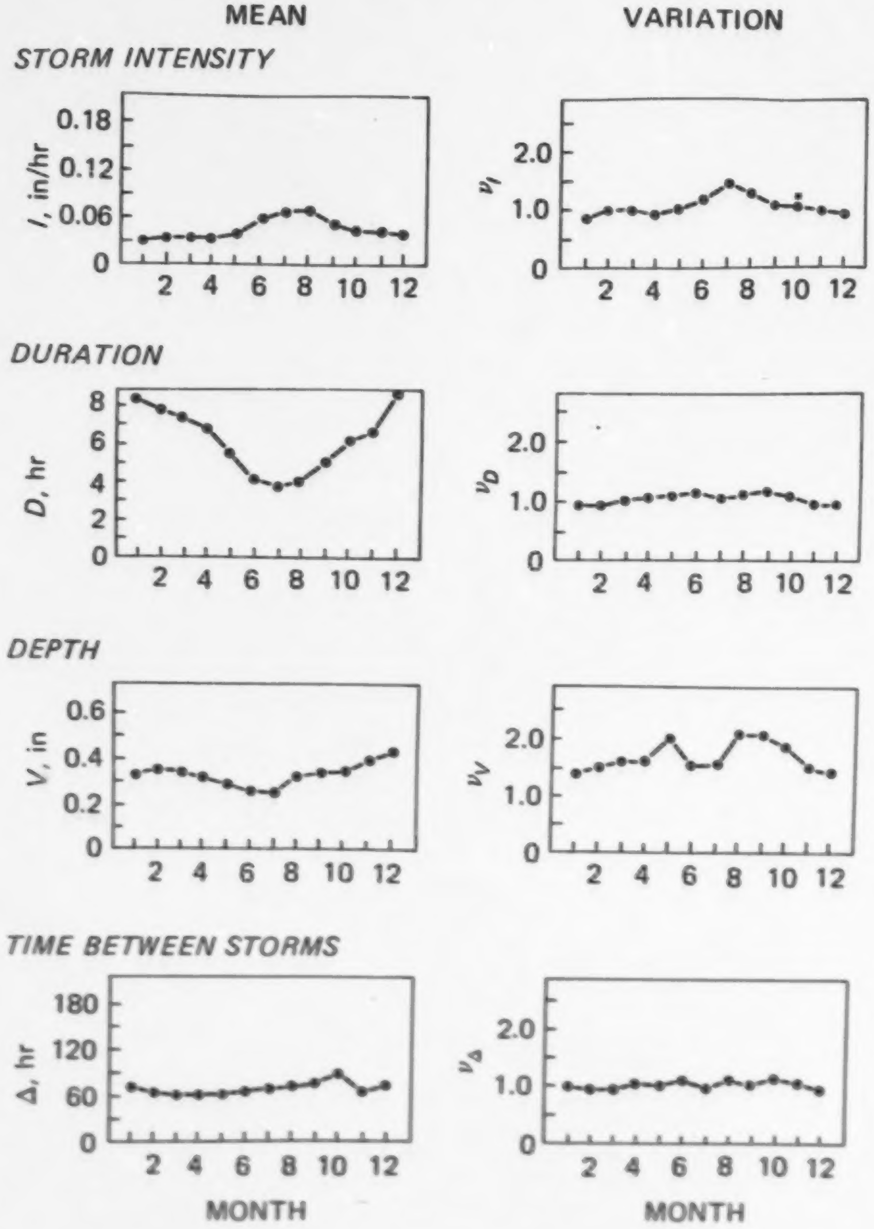


Figure 30. Monthly Statistical Characterization of Rainfall for Boston, Massachusetts

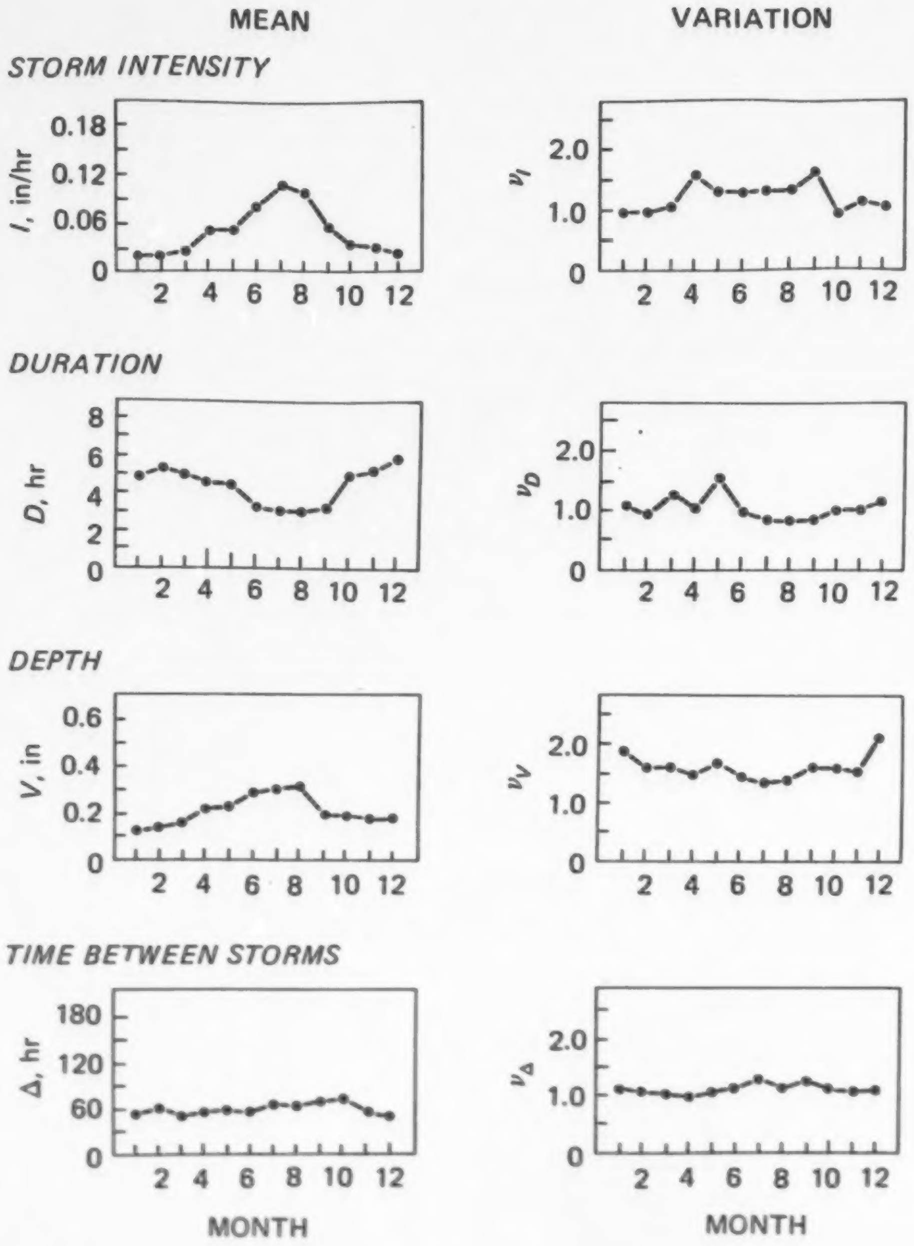


Figure 31. Monthly Statistical Characterization of Rainfall for Detroit, Michigan

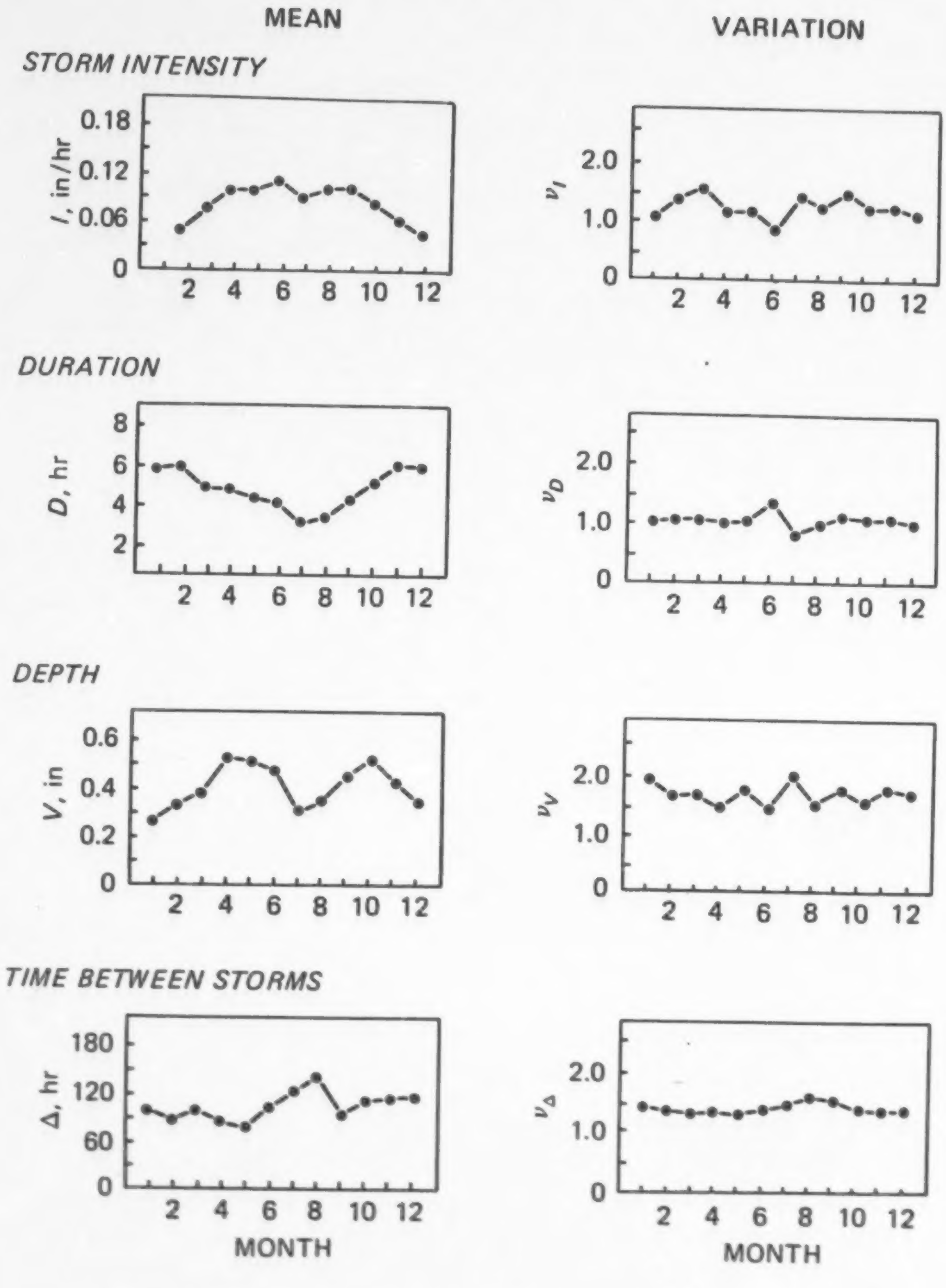


Figure 32. Monthly Statistical Characterization of Rainfall for Dallas, Texas

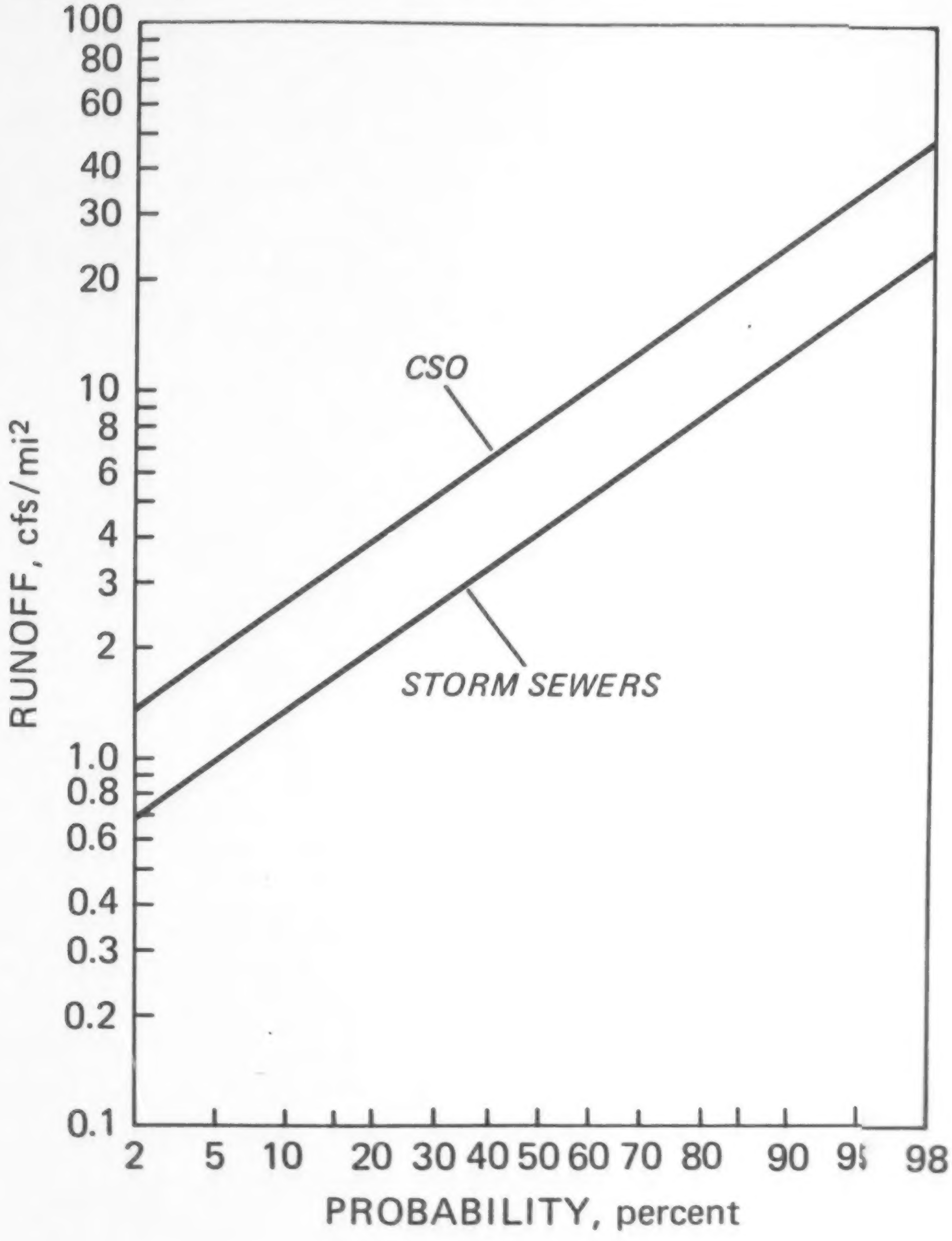


Figure 33. – Log Probability Distribution of Runoff for Boston

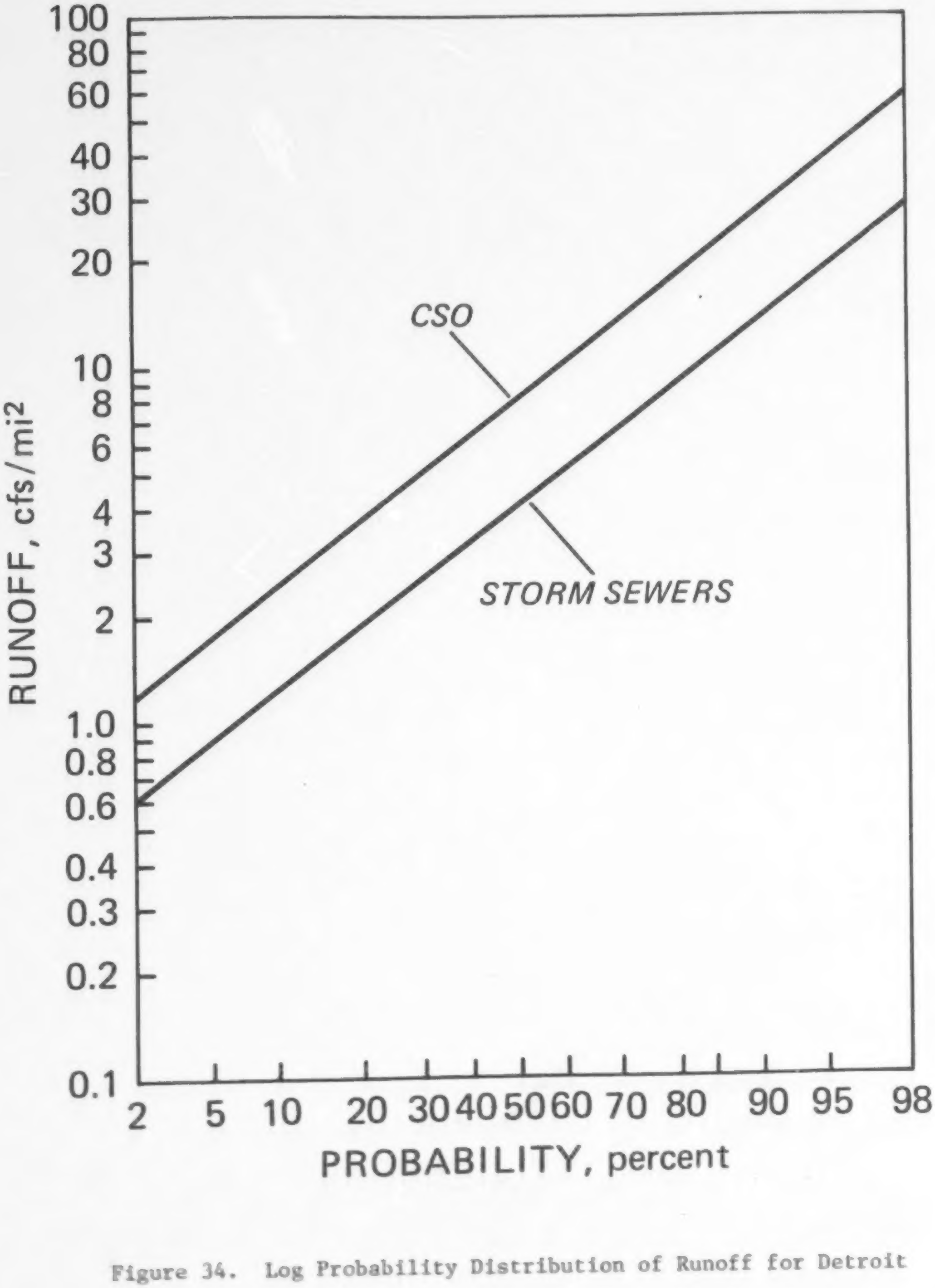


Figure 34. Log Probability Distribution of Runoff for Detroit

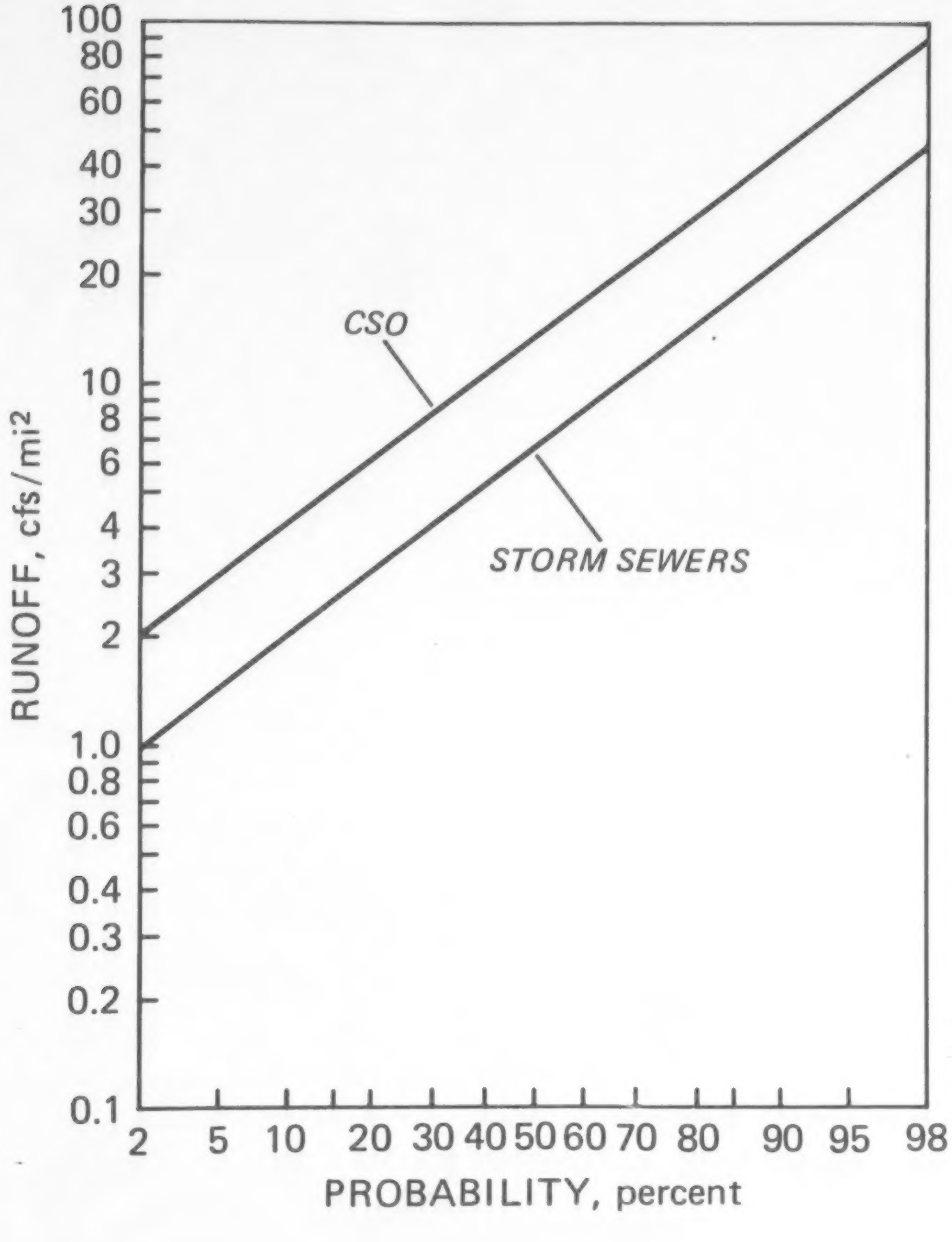


Figure 35. Log Probability Distribution of Runoff for Dallas

TABLE 13. RUNOFF (cfs/mi^2) FOR NINE SITES NATIONWIDE FOR CSO AND SEPARATE SEWER SYSTEMS

Area	Separate Sewers		CSO	
	50%	84%	50%	84%
Boston	4.13	9.78	8.26	19.56
New York	4.26	11.74	8.52	23.48
Dallas	6.71	17.07	13.42	34.15
Columbia	7.23	22.04	14.46	44.09
Portland	2.45	5.33	4.90	10.65
Tampa	8.91	23.27	17.81	46.51
Detroit	4.13	10.51	8.26	21.02
Salt Lake City	3.36	7.38	6.71	14.73
Denver	2.97	7.36	5.94	14.71

the calculations could be developed for each individual stream or stream segment of interest. The level of detail which could be considered is controlled by objective and resources. An example of the stream data utilized in the calculation for the Boston area is shown in figure 36 and the average regional stream flow data is presented in Table 14. For this analysis, the urban runoff concentrations were determined from the University of Florida data base (1) previously described.

The calculation procedure to determine the instream concentration at the outfall is that described in Table 3, steps 1 to 6. The analysis was conducted for concentrations of lead and ultimate oxygen demand (UOD) found in urban runoff and repeated for seven ratios of stream/urban area. An upstream concentration of zero was assumed. The results are summarized as in figure 37 for each drainage area ratio, constituent, location and type of sewer system. This specific example shows the probabilities of the resulting instream lead concentrations at the outfall of a CSO system located in the vicinity of Boston. For example, for a drainage area ratio of 20, there is a 50% probability of the concentration reaching 59 μg/ℓ at the outfall. As expected, the projected concentrations decrease as the drainage area ratio increases. This is due to the additional dilution available from the increased stream flow.

Results of the calcualtion can be summarized as shown (figure 38). The instream concentrations expected for a drainage area ratio of 10 are equal to or less than 580 μg/ℓ and 90 μg/ℓ, for the 90% and 50% probability levels. Similar plots for lead and UOD for each region are presented in the appendix.

For contaminants which are conservative or non-conservative, the maximum stream concentration caused by urban discharges will occur in the region of the discharges. Therefore the largest adverse effects from these discharges are anticipated to occur near the sources of urban overflows. The stream/river analysis can provide an approach which examines the situation at this critical location. All the contaminants in both CSO discharges and urban runoff can be examined by this technique. The one event related water quality problem which cannot be analyzed by this approach is dissolved oxygen since the reaeration rate varies with receiving water geometry. The procedures discussed in the section "Dissolved Oxygen Analysis of Rivers and Streams" which relates critical dissolved oxygen deficit, UOD, and stream purification ratio must supplement the calculations.

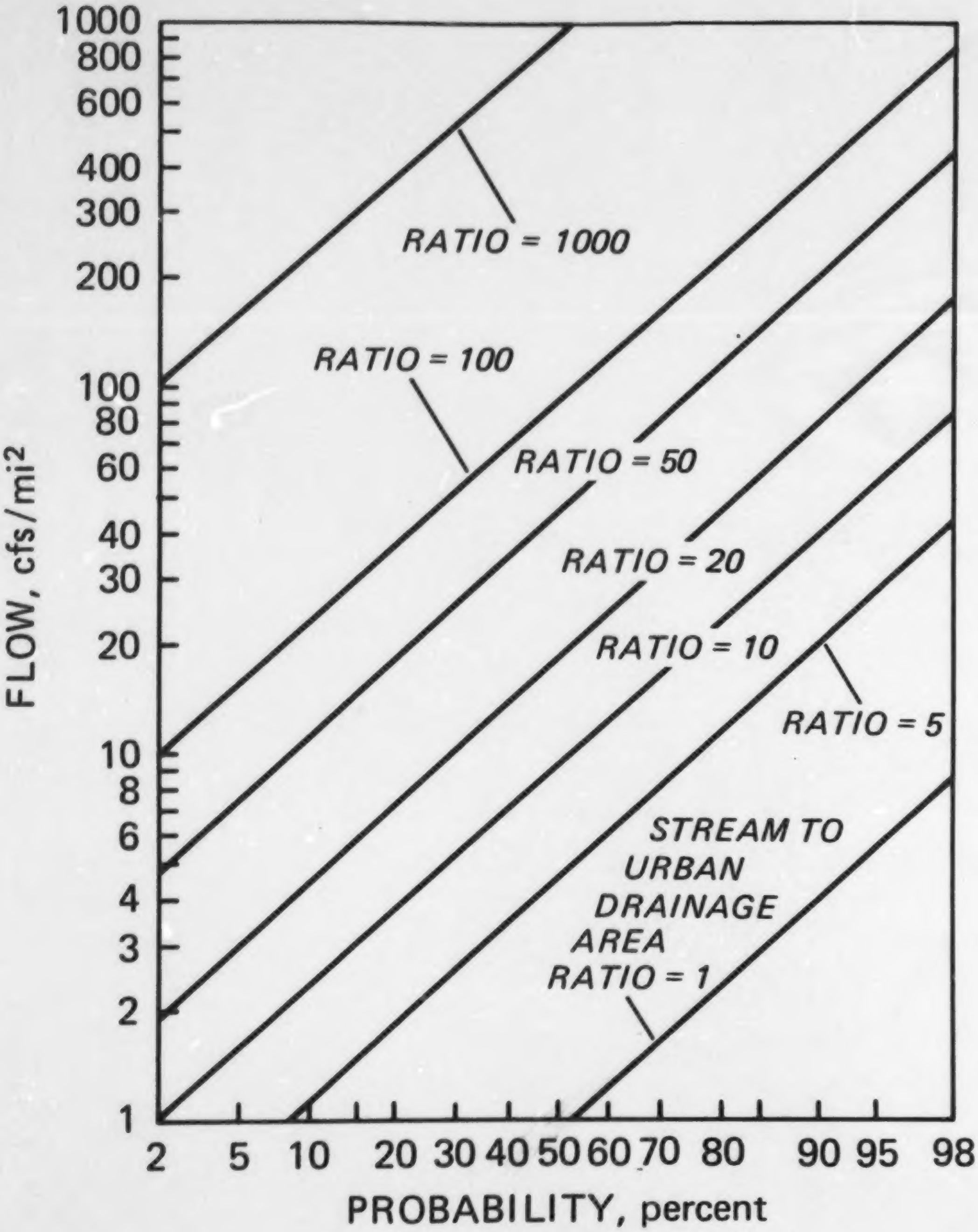


Figure 36. Log Probability Distribution of Stream Flow for Variable Stream to Urban Drainage Area Ratios

TABLE 14. STREAMFLOW (cfs/mi^2) FOR NINE REGIONAL LOCATIONS NATIONWIDE FOR A STREAM TO URBAN DRAINAGE AREA RATIO OF ONE

Area		50%	84%
Boston		0.931	2.76
New York		0.913	2.45
Dallas		0.364	1.71
Columbia		0.742	2.10
Portland(1)	a)	0.496	1.54
	b)	1.574	4.21
	c)	3.46	9.14
Tampa		1.12	2.65
Detroit		0.432	1.18
Salt Lake City		0.81	2.17
Denver		0.057	0.113

(1) Rivers in the Portland Ma. vicinity were divided into 3 categories due to the differences in mean flows.

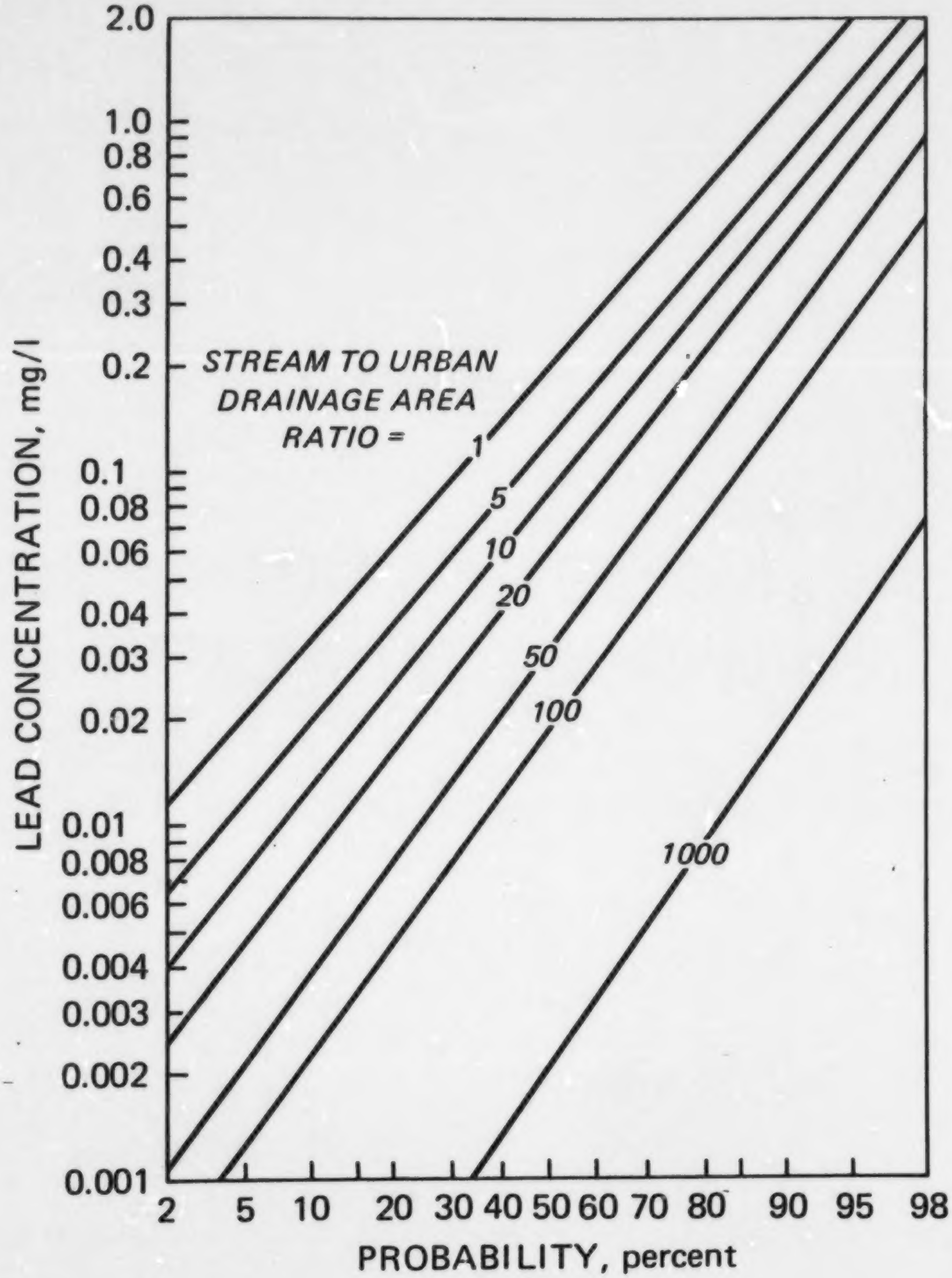


Figure 37. Log Probability Distribution of Instream Lead Concentration for Variable Stream to Urban Drainage Area Ratios from a CSO Discharge in Boston

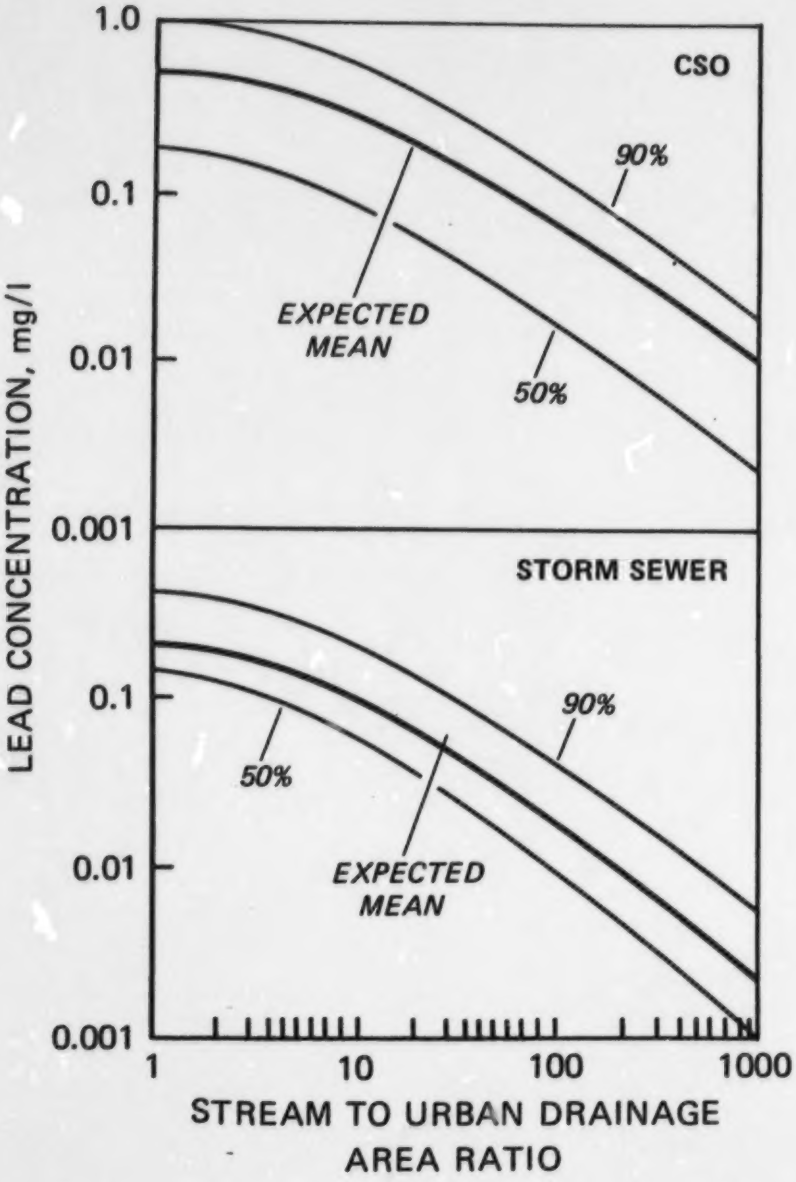


Figure 38. Instream Lead Concentrations Resulting from a CSO Discharge in Boston vs, Stream to Urban Drainage Area Ratios

SECTION 10

PROCEDURES FOR DEFINING WATER QUALITY PROBLEMS

Toxics

A technique for calculation of the event mean instream concentration probability distribution has been developed and example calculations are presented in the report section dealing with impacts in Rivers and Streams. Appendix B contains results for various regions of the country. These calculations employed pollutant loading information from the University of Florida data base (1) and appropriate information on regional stream flows and rainfall statistics. The calculation results consider a number of urban drainage area to stream drainage area ratios.

The technique presented has been used to calculate the event mean instream concentration probability distribution for lead, copper and ultimate oxygen demand. These three contaminants will be employed to illustrate one method of determining locations where water quality problems could be anticipated as a result of the discharges from CSO or urban runoff. Figures 39 and 40 present the event mean instream concentration of lead (for CSO and urban runoff) vs. probability for stream flows and rainfalls characteristic of the southwest region (Dallas). Results are shown for stream to urban drainage area ratios ranging from one to one thousand. The lower drainage area ratio (one) represents undiluted CSO and/or urban runoff. The calculated instream concentrations can be plotted against drainage area ratio for a constant probability. This transformation is illustrated in figure 41 for the median, expected mean and 90% probability levels.

Water Quality Criteria can also be plotted on these figures as shown. The criteria plotted are the maximum concentration of lead promulgated by EPA and the lead concentration which would result in 50% fish mortality during a 12 hour storm. This concentration calculation utilized the techniques developed in this project and consider the carryover effects from average storms for both CSO and urban runoff. The carryover effects are for undiluted runoff.

Figures 39 and 40 can be employed to determine what percentage of rainfall events will result in instream lead concentrations in excess of water quality criteria. Figure 41 can be employed to determine the drainage area ratio for which water quality criteria will not be violated a given percentage of time during periods of rainfall.

Examination of these 3 figures indicate that the water quality criteria determine, to a very large extent, if wate quality problems are to be anticipated from CSO and urban runoff discharges. The EPA maximum criteria

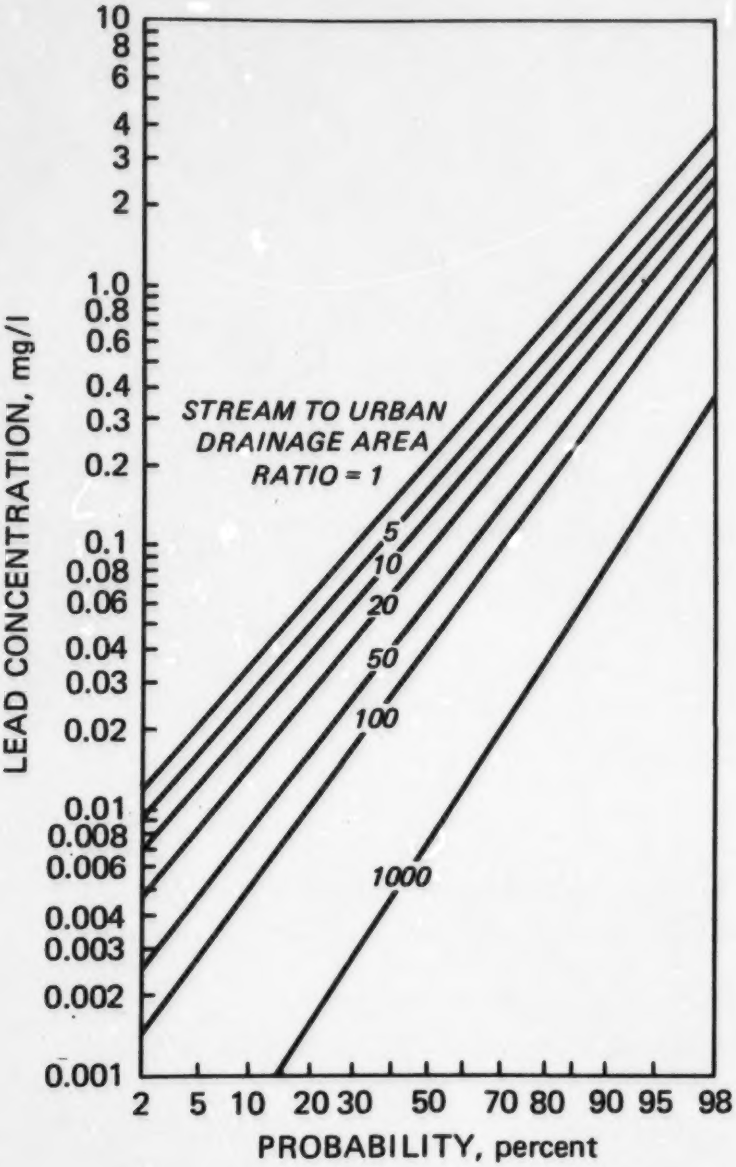


Figure 39. Log Probability Distribution of Instream Lead Concentration for Variable Stream to Urban Drainage Area Ratios from a CSO Discharge in Dallas

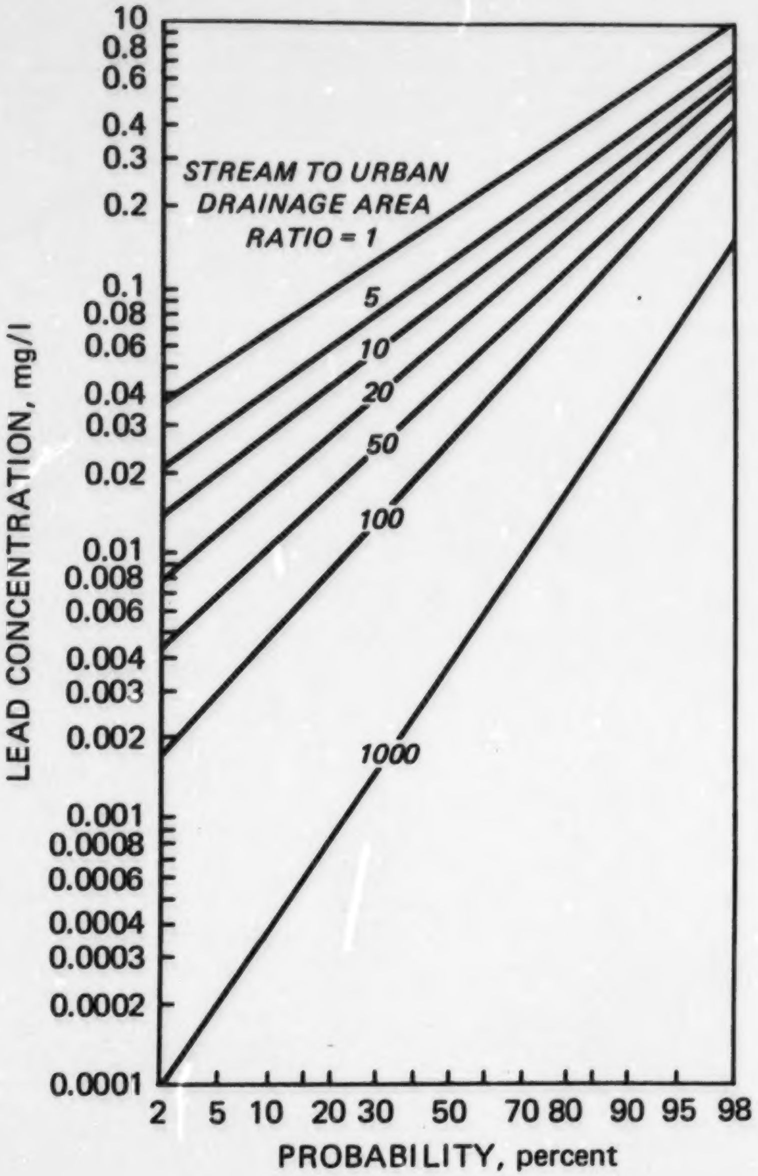


Figure 40. Log Probability Distribution of Instream Lead Concentration for Variable Stream to Urban Drainage Area Ratios from a Storm Sewer in Dallas

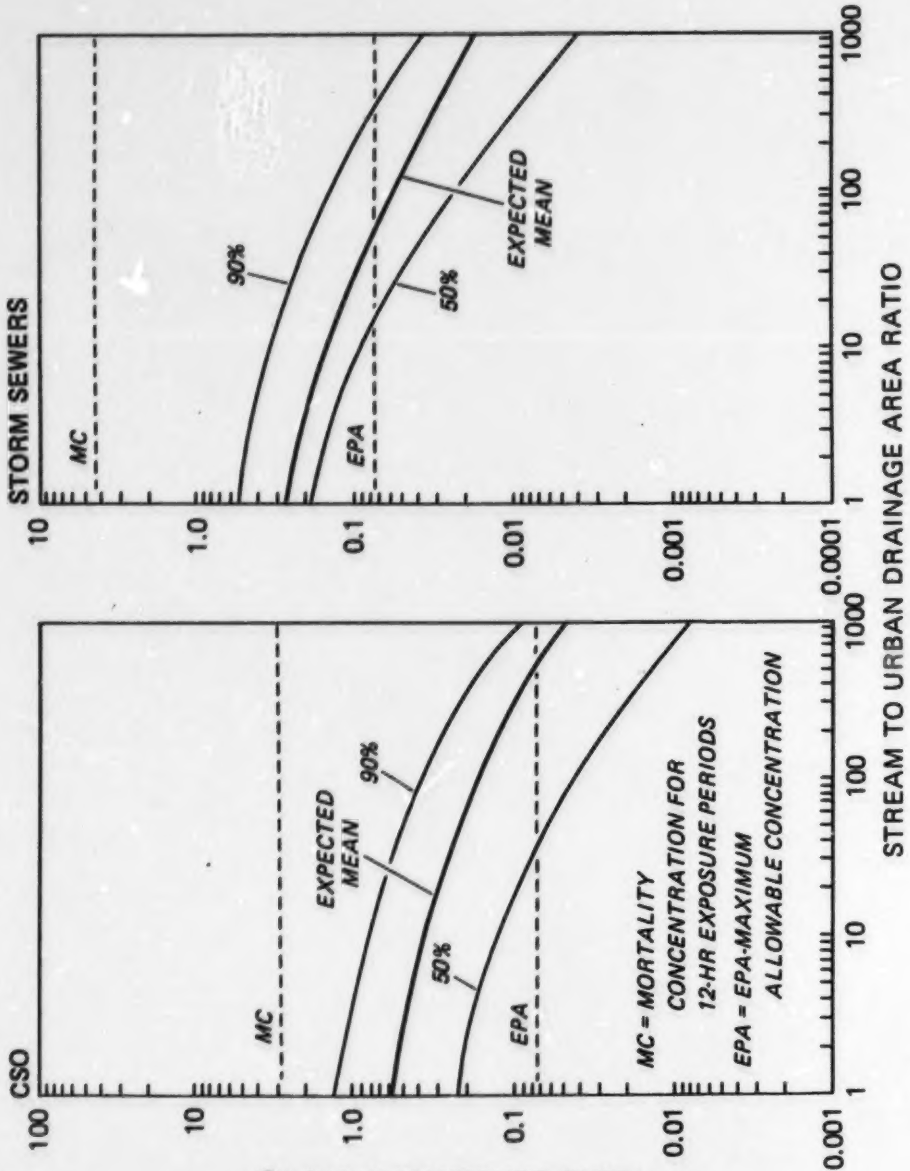


Figure 41. Instream Lead Concentration Resulting from CSO and Storm Sewer Discharges in Dallas vs. Stream to Urban Drainage Area Ratios

for lead is violated at the 90% probability for large streams with small urban areas considering both CSO and urban runoff. By contrast if water quality criteria are considered which reflect the time scales associated with storm events (12 hour mortality concentration) there are relatively large safety margins at the 90% probability level and water quality problems should not be anticipated from the lead contained in CSO and/or urban runoff. Neither numerical criteria used on this figure is completely satisfactory as discussed in the criteria section of this report.

A second significant observation may be obtained from examination of figure 42 for the New York region; which deals with the effects of drainage area ratio on the anticipation of water quality problems. In this region of the country the ratio of stream to urban drainage divides the sites at which water quality problems can be anticipated. Considering the EPA maximum allowable concentration water quality problems would be anticipated for stream to urban drainage area ratios below on the order of 100 to 1. For drainage area ratios above this level water quality criteria are not violated. Additional computational results are presented in figures 43, 44, 45 and 46 for other regions of the country and for copper.

Table 15 presents the drainage area ratio required to meet water quality criteria for lead in several regions of the country.

Dissolved Oxygen

A similar analysis and calculation procedure has been developed considering ultimate oxygen demand and dissolved oxygen. The procedure for calculating the probability distribution of the UOD instream concentrations is identical to that employed for toxics. The results of these calculations are presented on figures 47 to 52 for various regions of the country.

In addition, calculations discussed in this report under dissolved oxygen analysis in streams and rivers are required to relate UOD to stream dissolved oxygen criteria. Two dissolved oxygen criteria are considered. The more stringent criteria is the usual 5 mg/ℓ. A second criteria is for low dissolved oxygen levels that should not result in fish mortality for the time scales associated with runoff from storm events. This lower criteria was calculated using the time variable mortality analysis for 12 hour exposure periods and average values of $\frac{R_D}{k_T}$ and k_R (1.99 and 1.152, respectively).

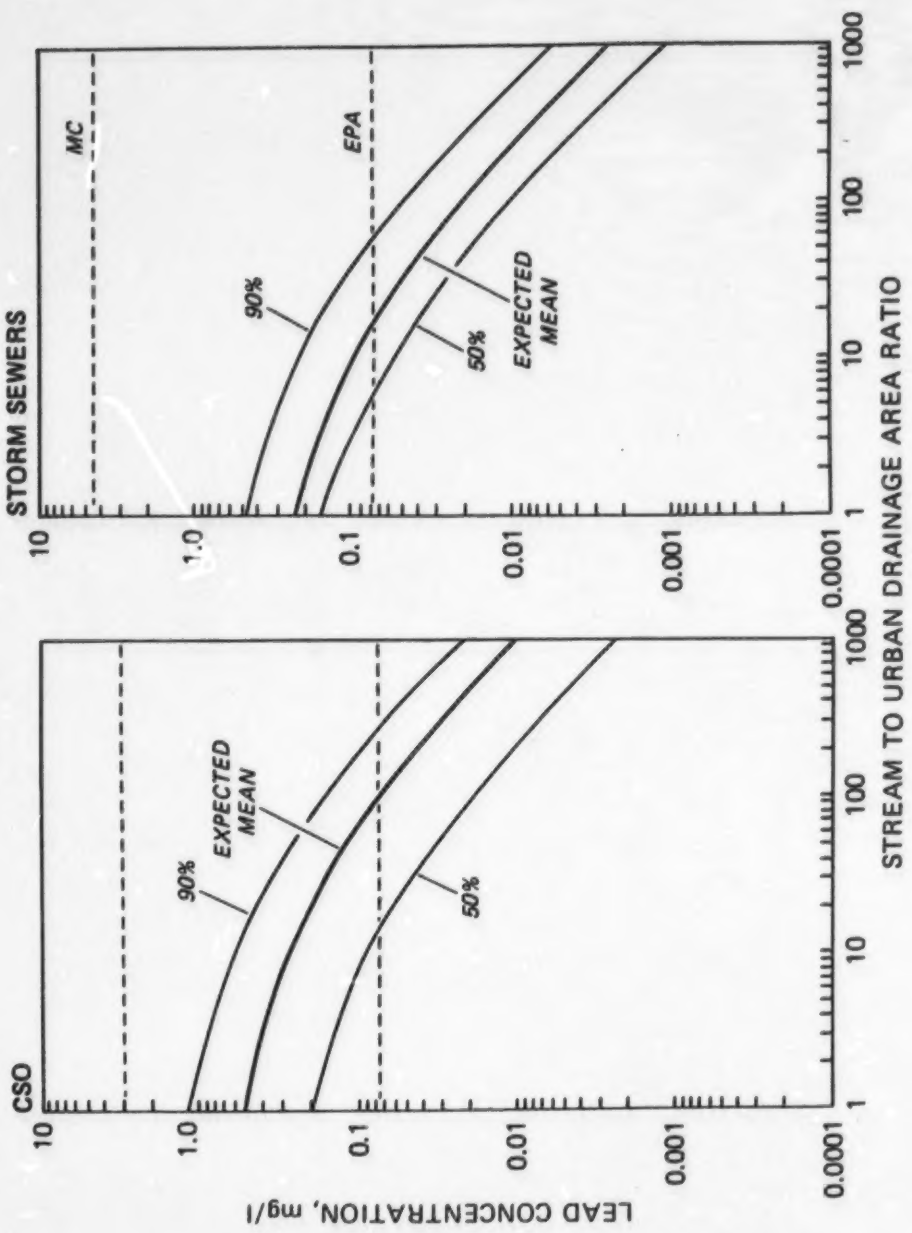


Figure 42. Instream Lead Concentration Resulting from CSO and Storm Sewer Discharges in New York vs. Stream to Urban Drainage Area Ratios

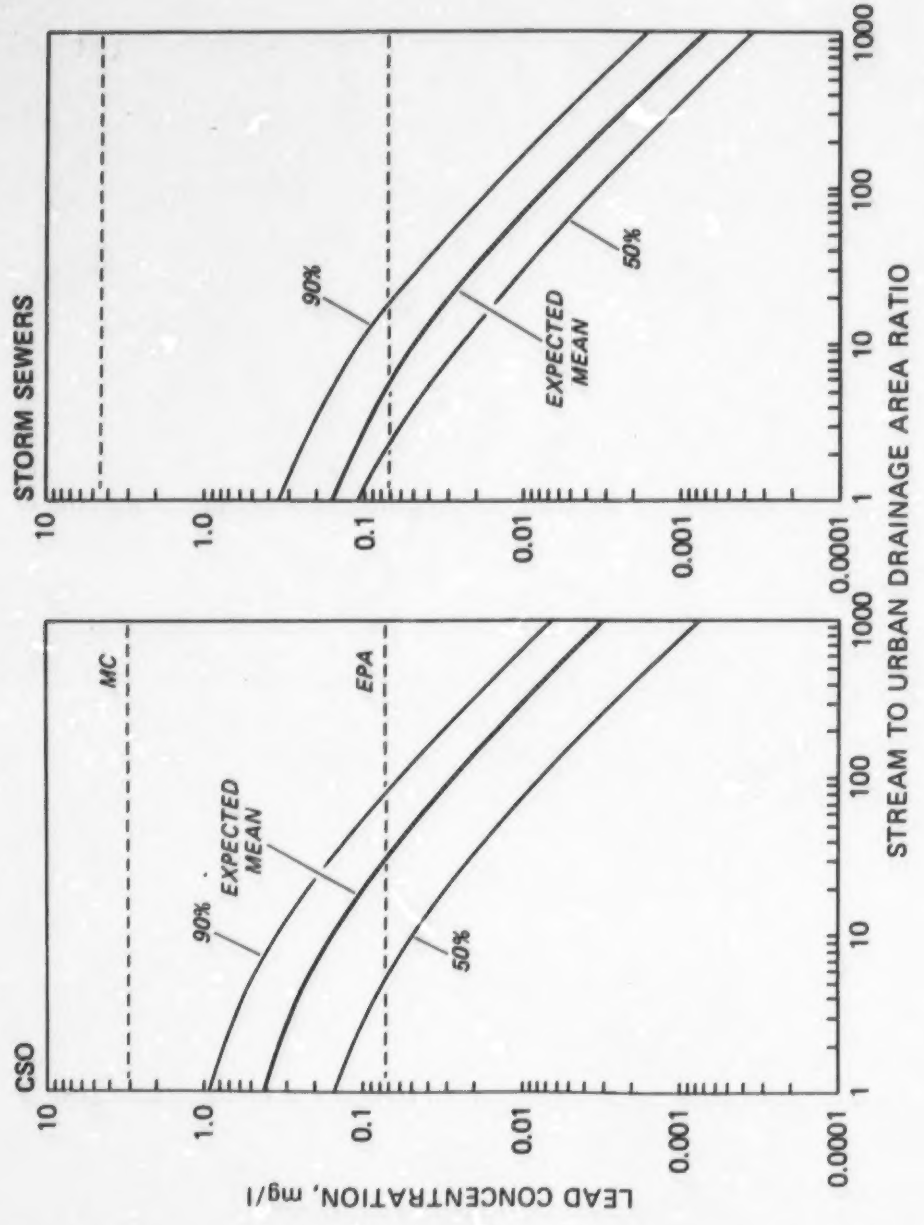


Figure 43. Instream Lead Concentration Resulting from CSO and Storm Sewer Discharges in Portland vs. Stream to Urban Drainage Area Ratios

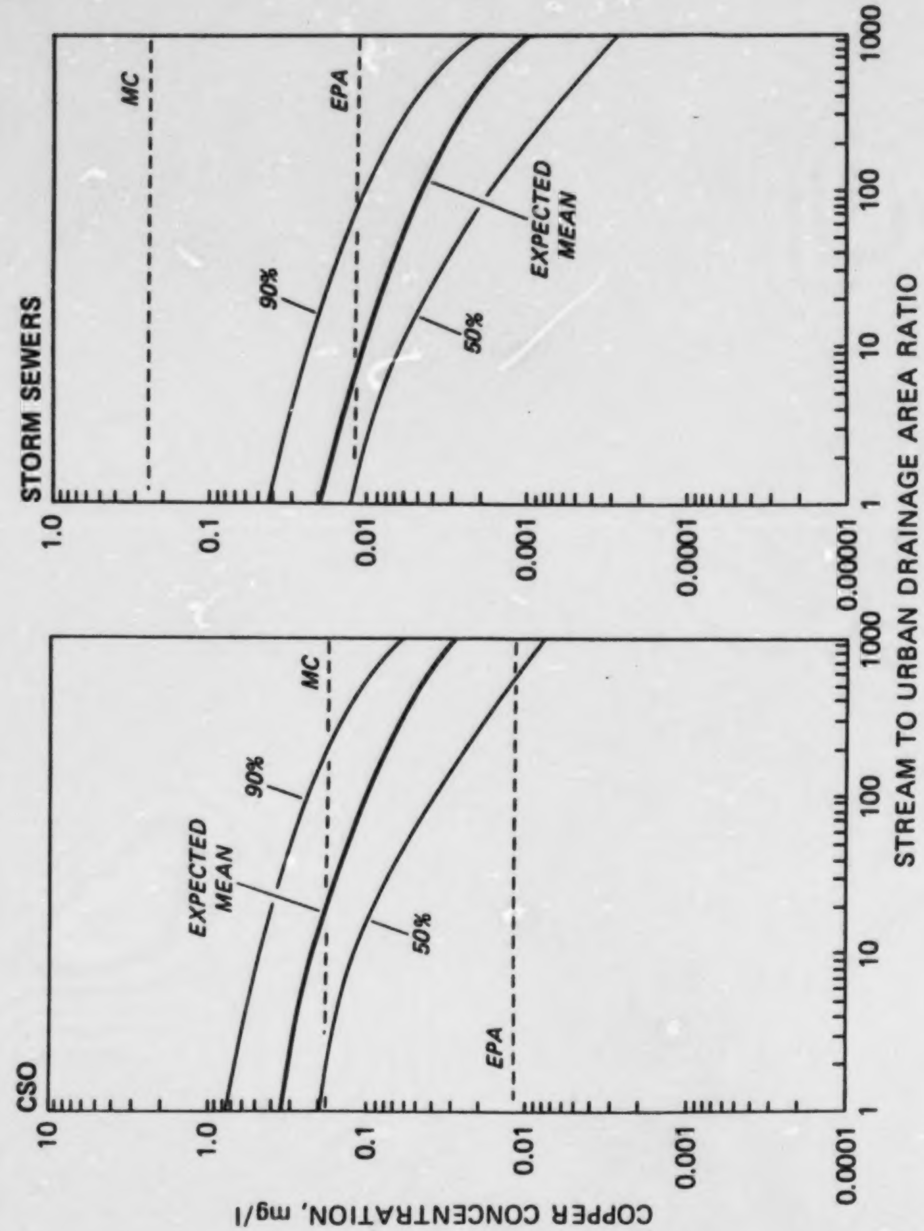


Figure 44. Instream Copper Concentration Resulting from CSO and Storm Sewer Discharges in Dallas vs. Stream to Urban Drainage Area Ratios

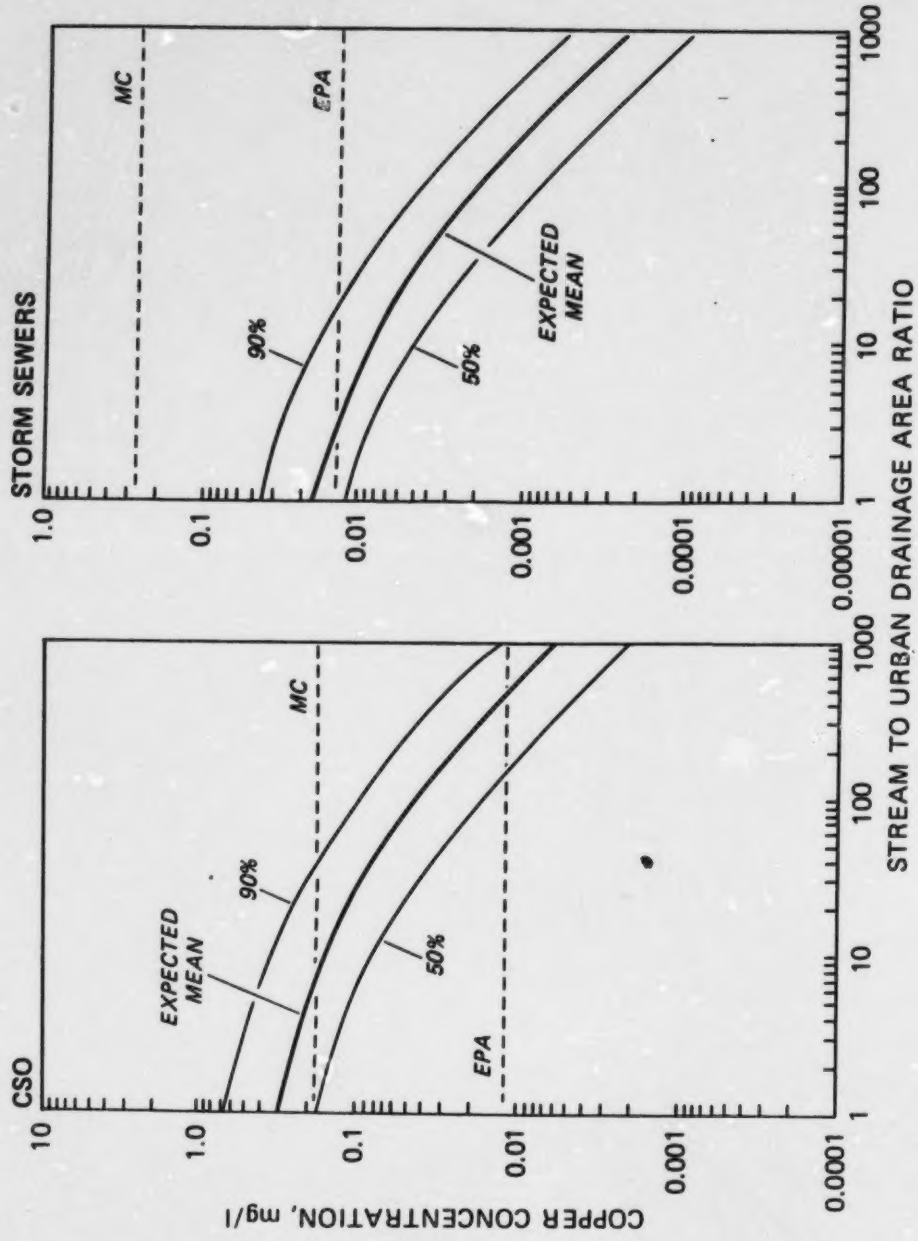


Figure 45. Instream Copper Concentration Resulting from CSO And Storm Sewer Discharges in New York vs. Stream to Urban Drainage Area Ratios

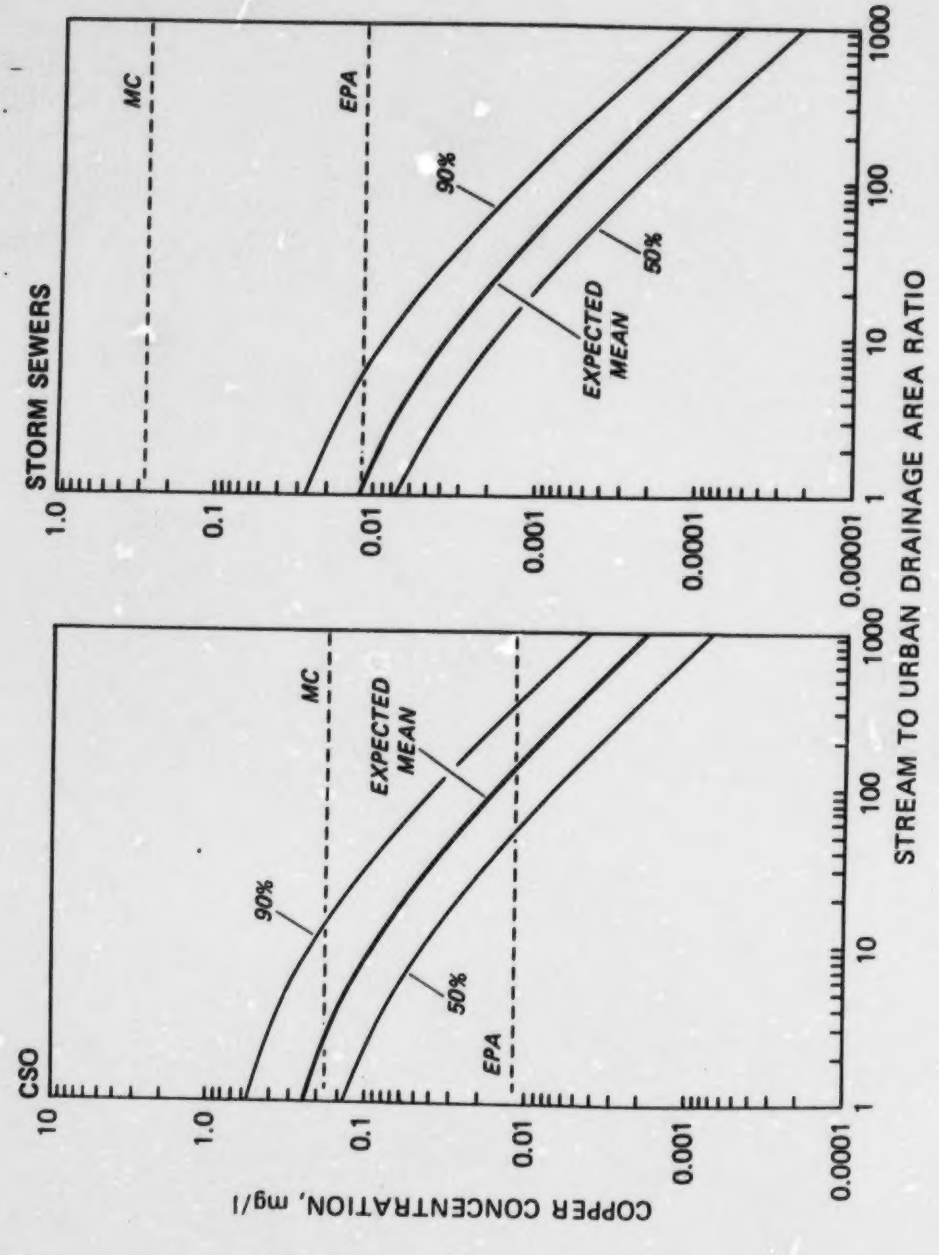


Figure 46. Instream Copper Concentration Resulting from CSO and Storm Sewer Discharges in Portland vs. Stream to Urban Drainage Area Ratios

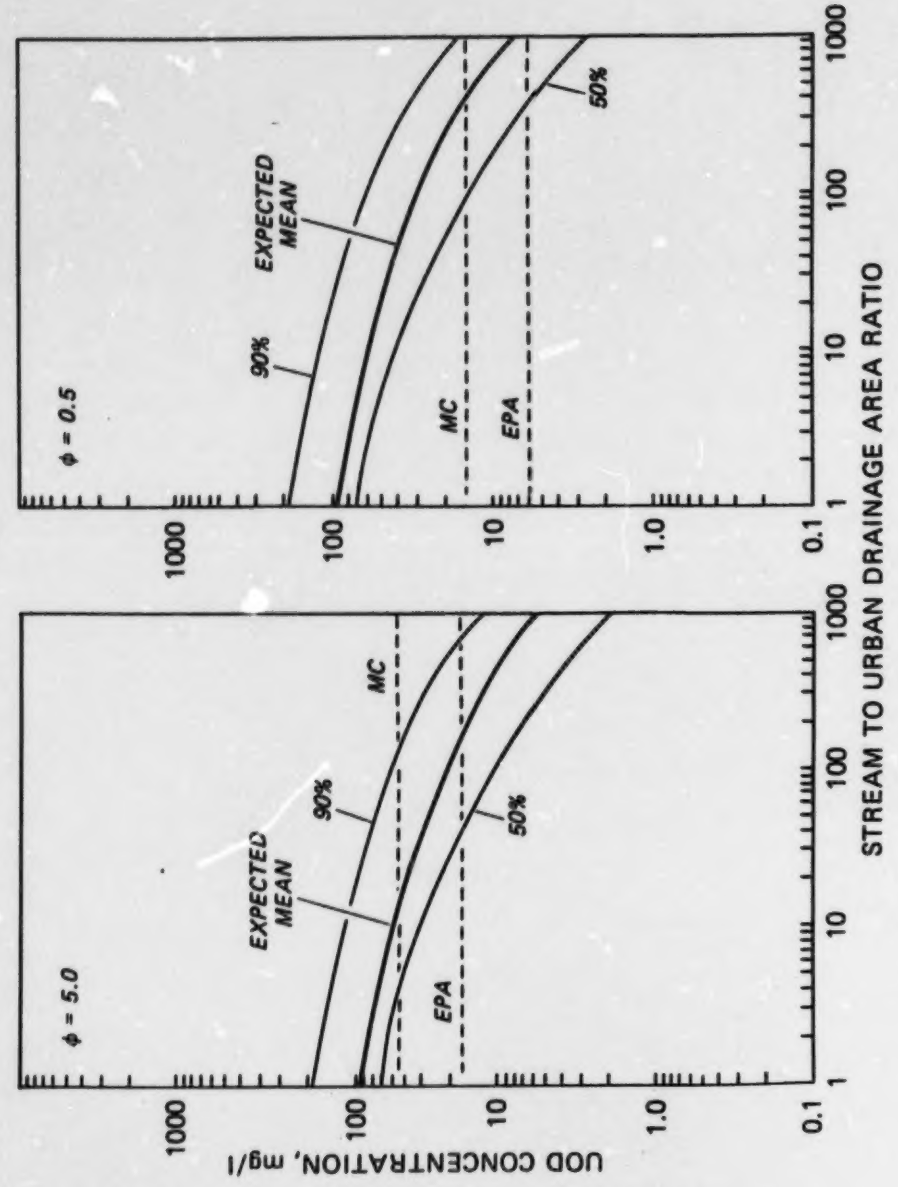


Figure 47. Instream UOD Concentration Resulting from CSO Discharges in Dallas vs. Stream to Urban Drainage Area Ratios

TABLE 15. STREAM/URBAN DRAINAGE AREA RATIO REQUIRED TO MEET THE EPA MAXIMUM STANDARD (0.074 mg/ℓ) FOR LEAD AT 90% AND 98% PROBABILITY

	Separate Sewers		CSO	
Region	98%	90%	98%	90%
Boston				
New York	250	50	1000	240
Dallas	>1000	300	>1000	>1000
Columbia	400	100	>1000	450
Tampa	325	85	>1000	360
Detroit	325	110	>1000	450
Salt Lake City	150	53	900	200
Denver	>1000	600	>1000	>1000
Portland (1)	200	65	1000	250
(2)	50	18	325	70
(3)	23	8	130	30

Note: The resulting concentrations for a drainage area ratio of one was sufficient to meet the time variable mortality analysis standard for a 4.5 hour exposure.

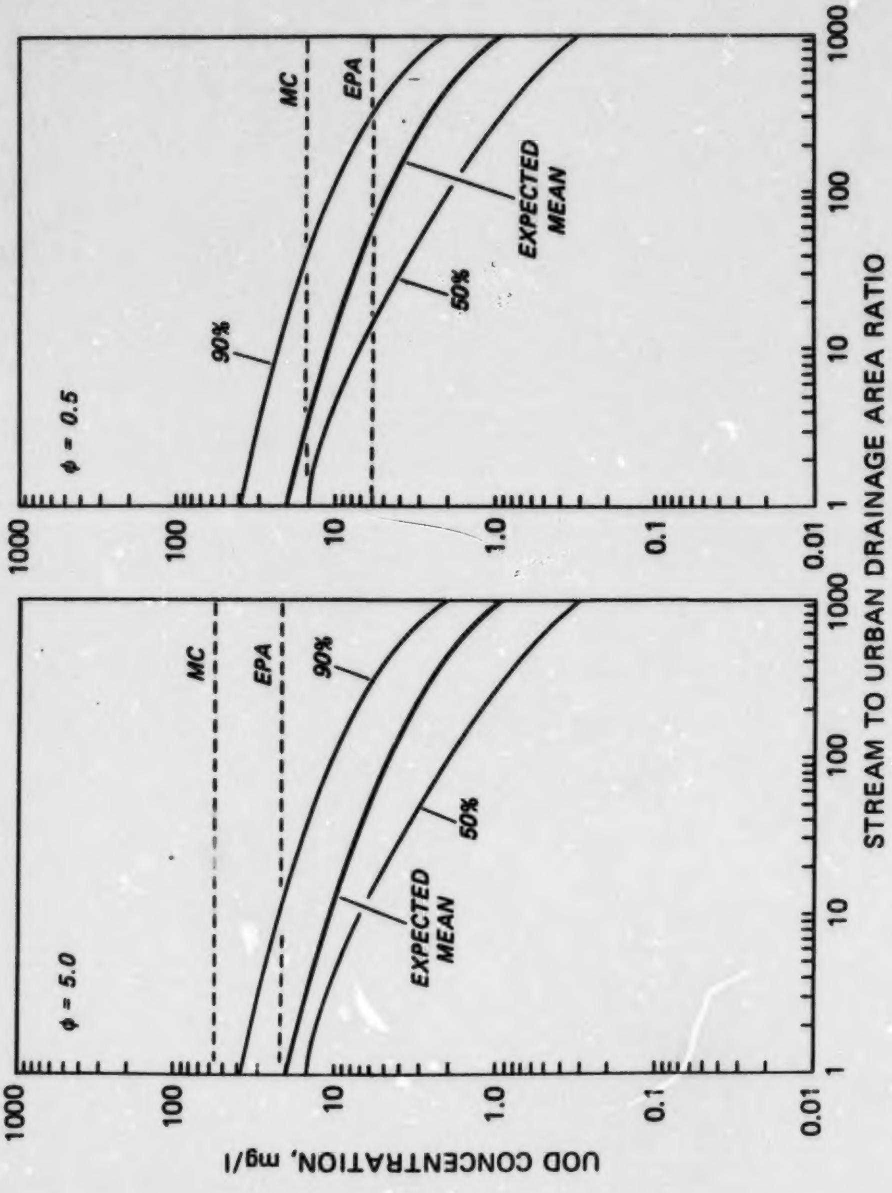


Figure 48. Instream UOD Concentration Resulting from Storm Sewer Discharges in Dallas vs. Stream to Urban Drainage Area Ratios

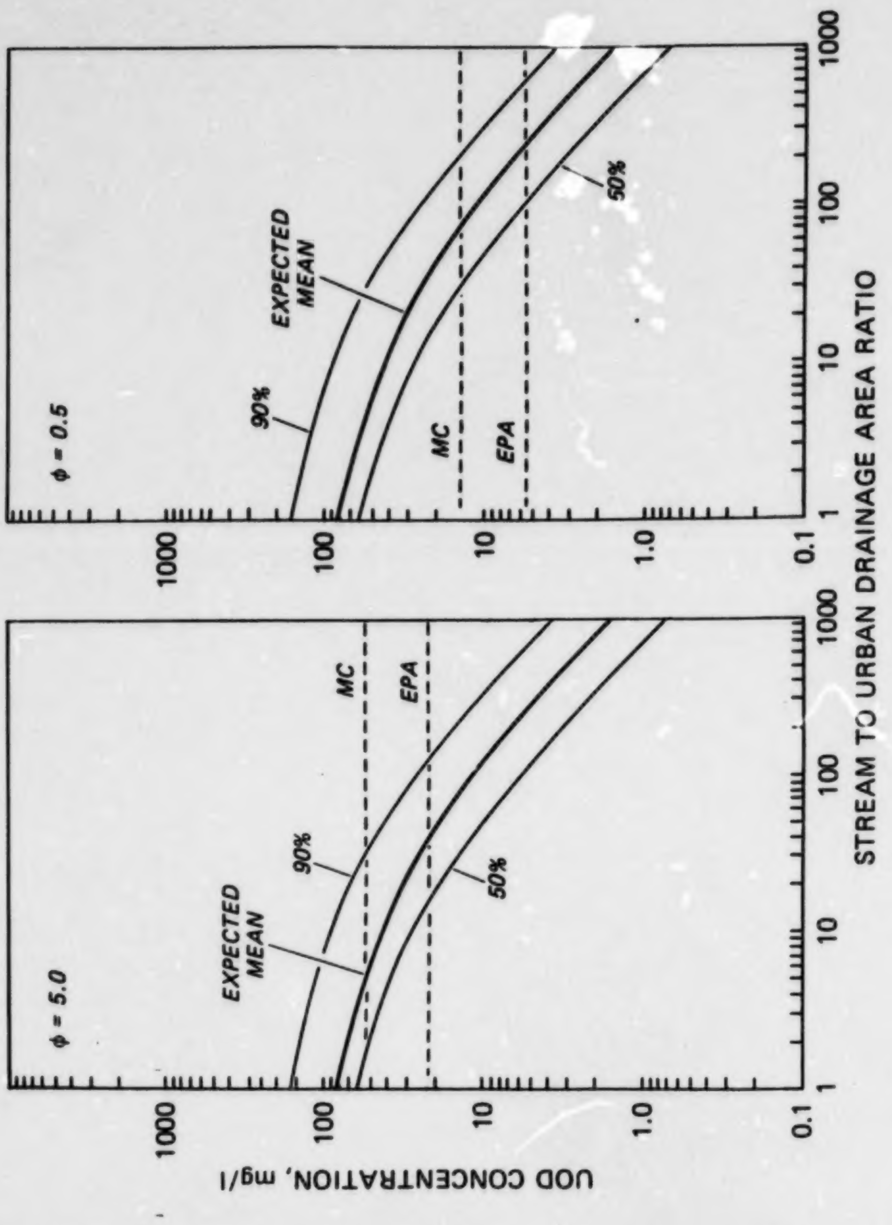


Figure 49. Instream UOD Concentration Resulting from CSO Discharges in New York vs. Stream to Urban Drainage Area Ratios

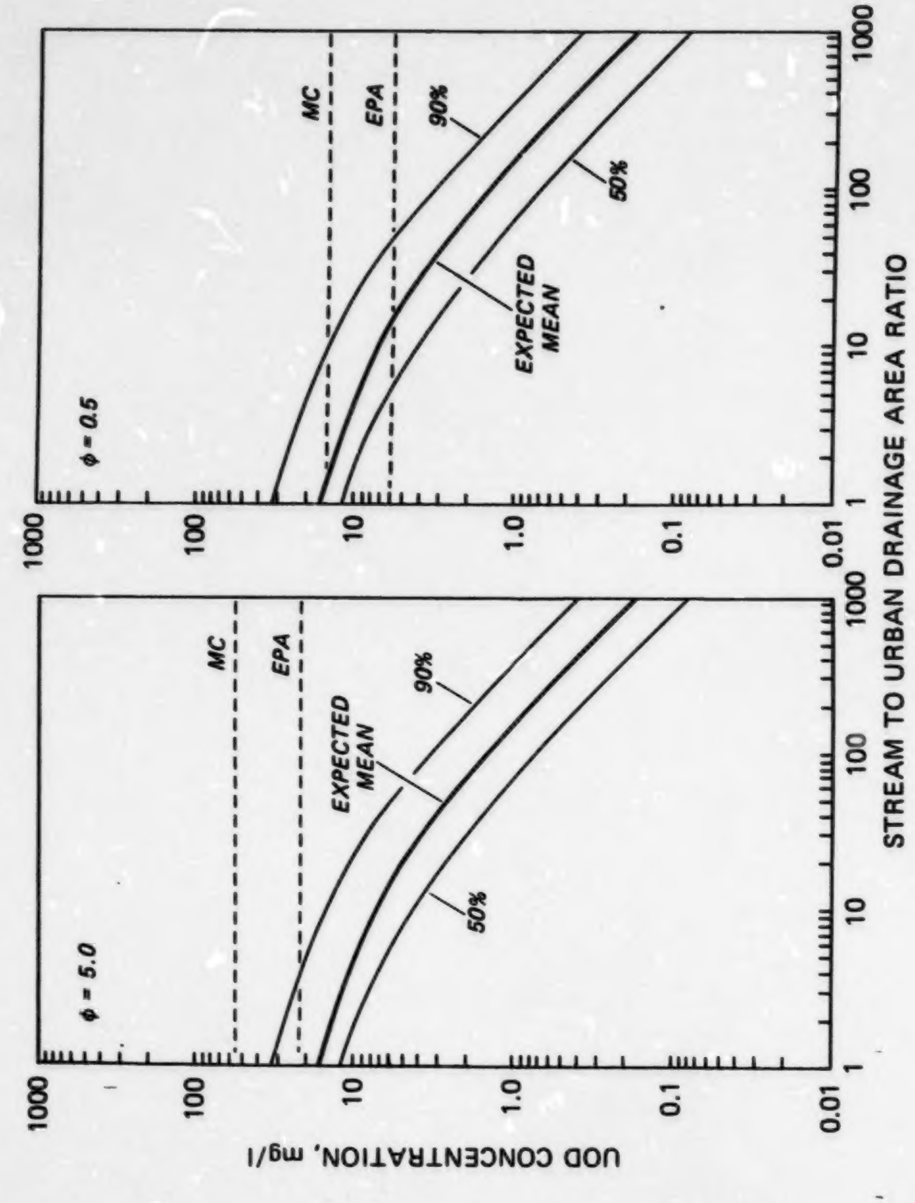


Figure 50. Instream UOD Concentration Resulting from Storm Sewer Discharges in New York vs. Stream to Urban Drainage Area Ratios

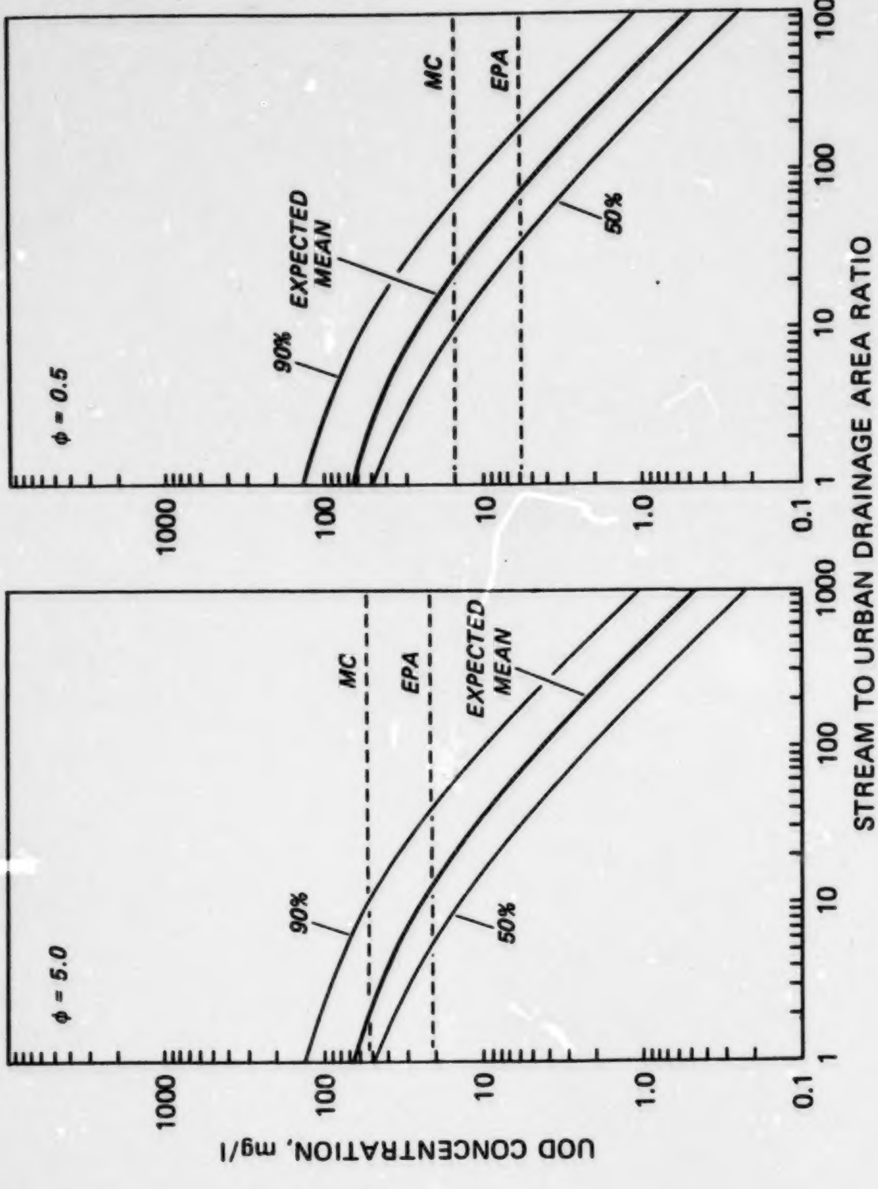


Figure 51. Instream UOD Concentration Resulting from CSO Discharges in Portland vs. Stream to Urban Drainage Area Ratios

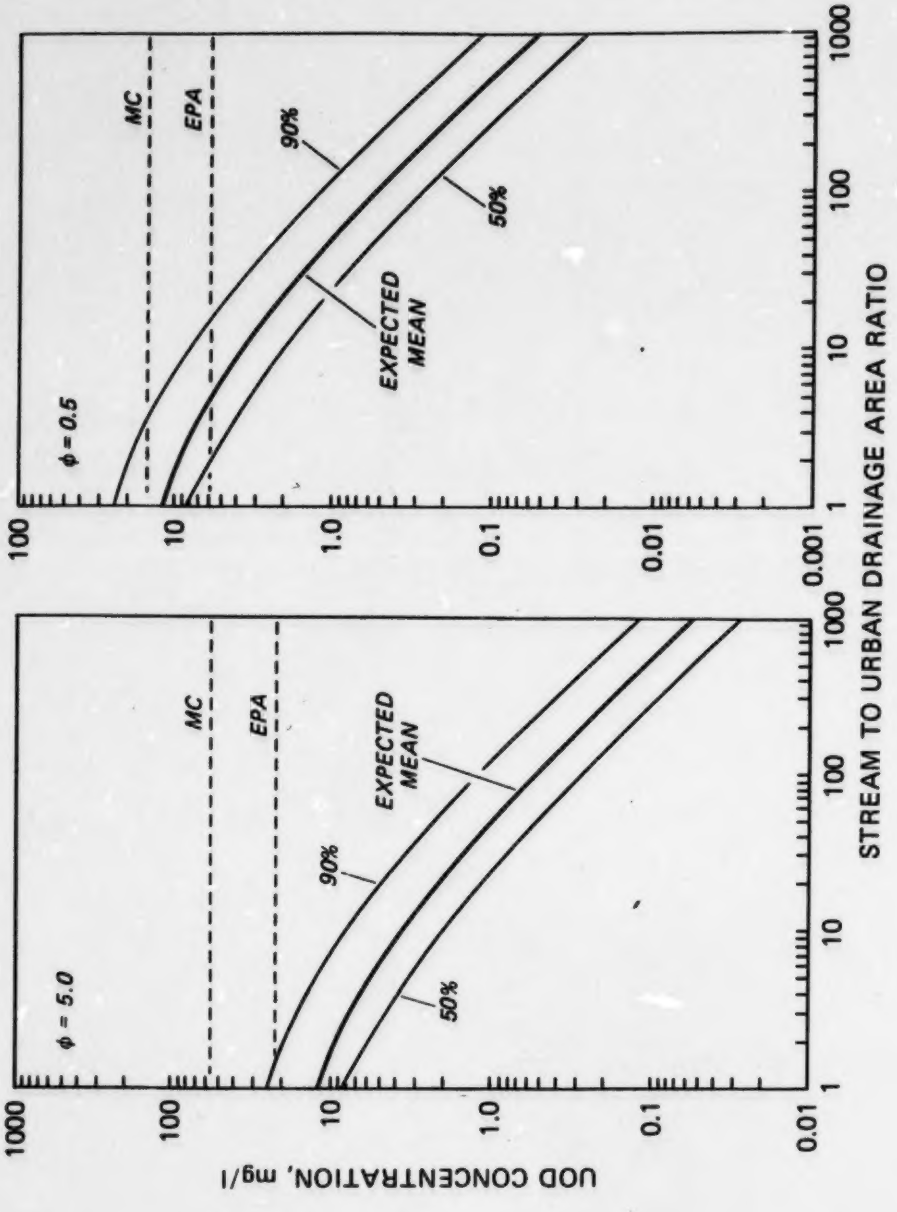


Figure 52. Instream UOD Concentration Resulting from Storm Sewer Discharges in Portland vs. Stream to Urban Drainage Area Ratios

The transformation of oxygen standards into allowable UOD stream concentrations employed equation (27). The calculations were done for two purification ratios (0.5 and 5.0), assuming a saturation value of 8 mg/ℓ and also assuming that all allowable deficit is from the UOD discharge. For example, if the allowable oxygen concentration is 0.434 then the deficit, D_C, equals 7.566. For a purification ratio of 0.5, the maximum UOD discharge concentration that will just maintain the oxygen standard equals:

$$L_o = \frac{D_C}{\phi^{\frac{\phi}{1-\phi}}} = \frac{7.566}{0.5^{\frac{0.5}{1-0.5}}} = 15.132$$

If there are other sources of deficit, then the D_C value in eq. (27) is decreased to account for the additional sink.

The dissolved oxygen criteria and corresponding UOD concentrations are summarized in Table 16 and plotted on figures 47-52. The relative importance of the criteria level and the drainage area ratio can be observed on these figures.

TABLE 16. DISSOLVED OXYGEN CRITERIA AND CORRESPONDING UOD CONCENTRATIONS AT THE DISCHARGE SITE TO PREVENT FISH MORTALITY

	D.O. (mg/ℓ)	UOD (mg/ℓ)	
		$\phi = 0.5$	$\phi = 5.0$
EPA criteria	5.0	6.0	22.4
Minimum D.O. for no mortality at 12 hours	0.43	15.13	56.55

The examples and figures presented in this section of the report do not consider background instream concentrations. As discussed the calculation procedure can accommodate data on upstream concentrations. It is also possible to consider adjustment of criteria to account for background concentrations.

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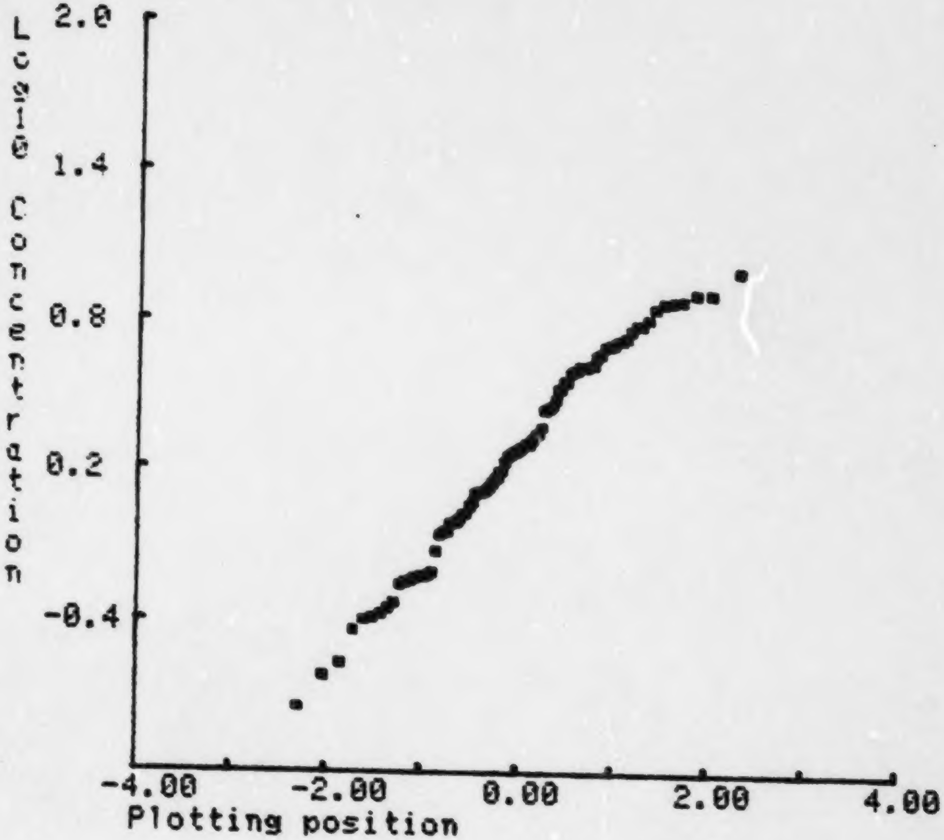
APPENDIX

a) Log-normal probability plots of contaminants from the University of Florida data base.

b) Calculated Instream Contaminant Concentrations (Lead - UOD) vs. Stream/Urban Drainage Areas in different regions of the country.

AMMONIA----COMBINED SEWER SYSTEM



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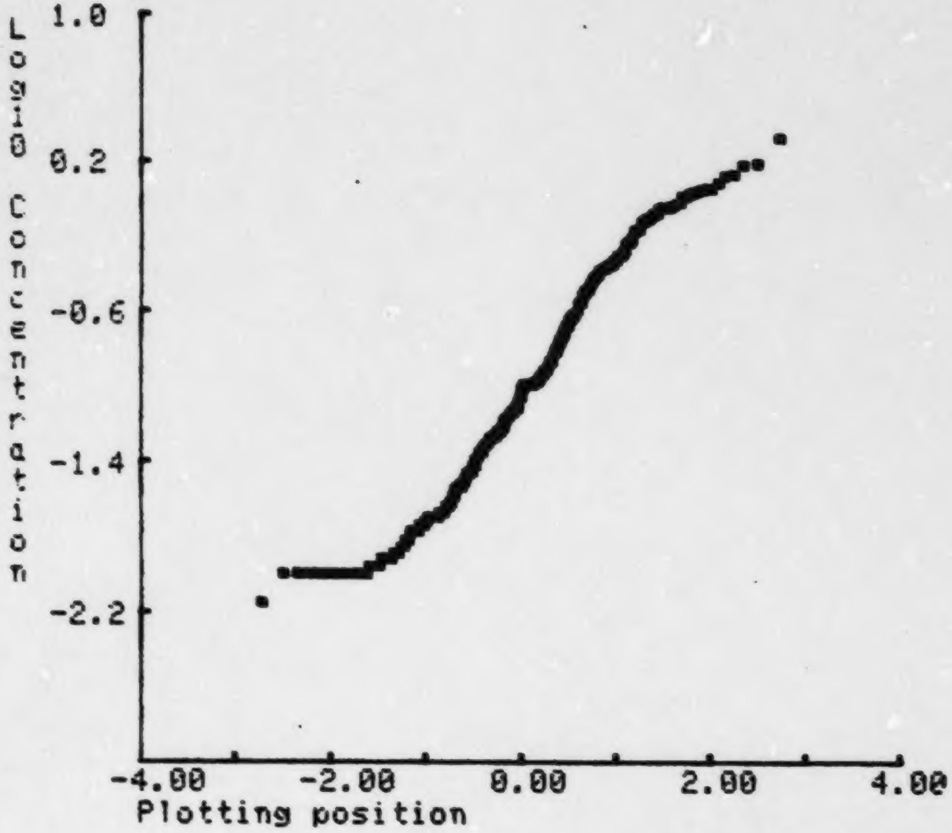
AMMONIA---STORM SEWER SYSTEM



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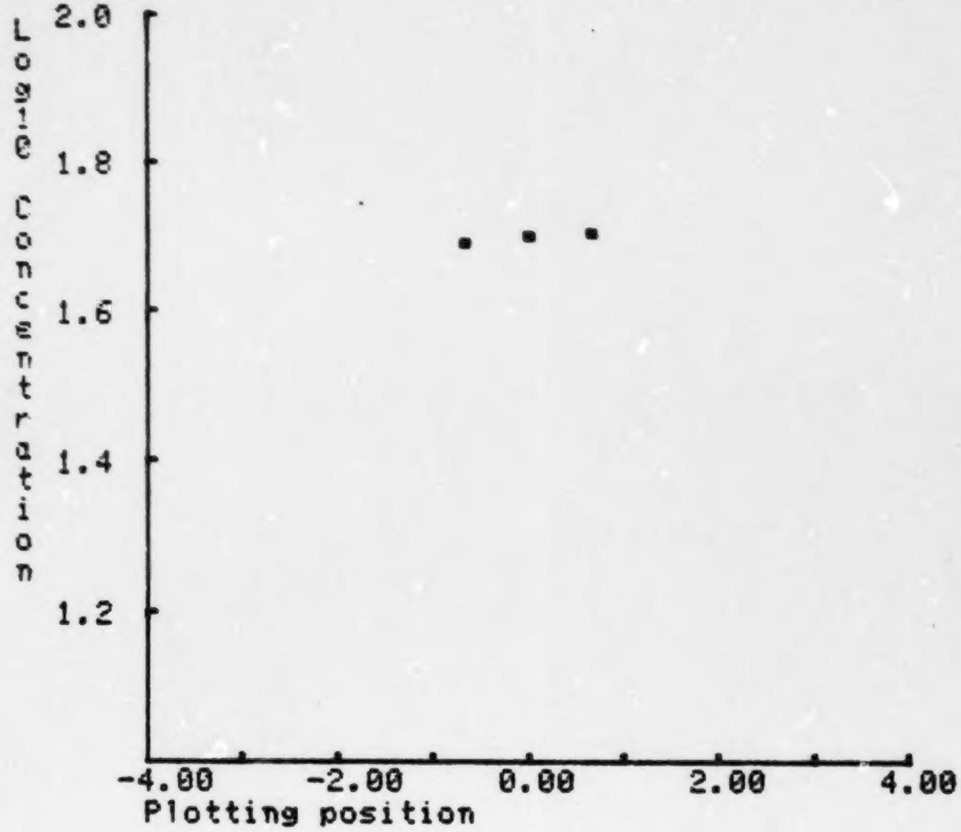
ARSENIC-----COMBINED SEWER SYSTEM



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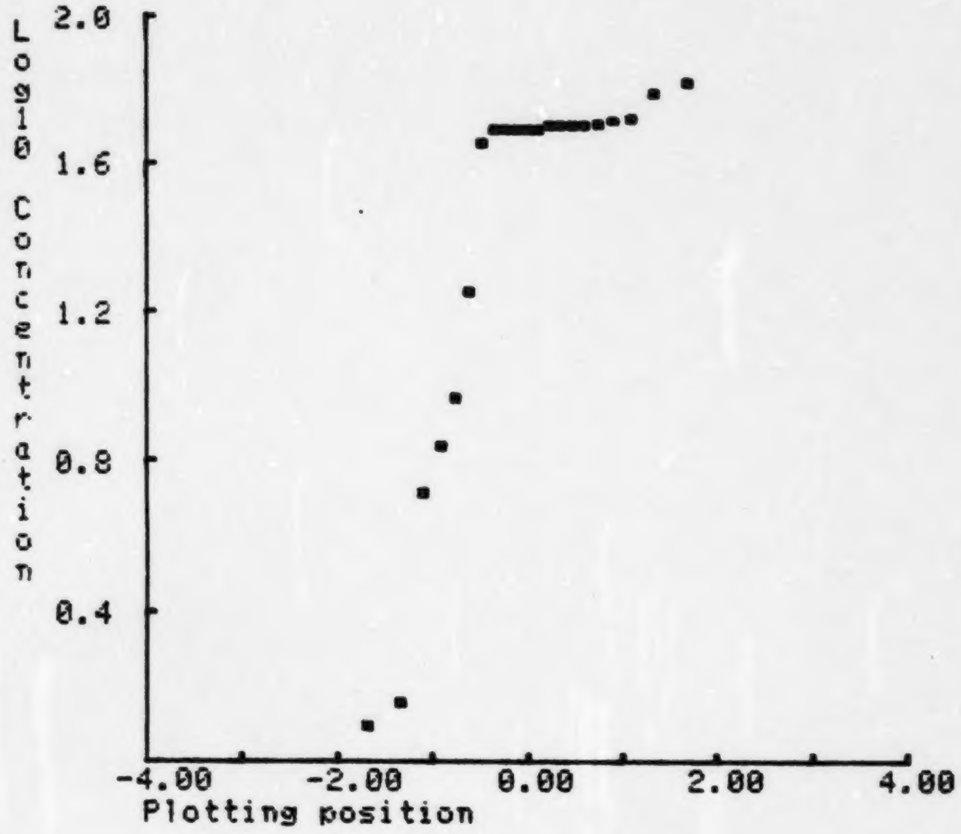

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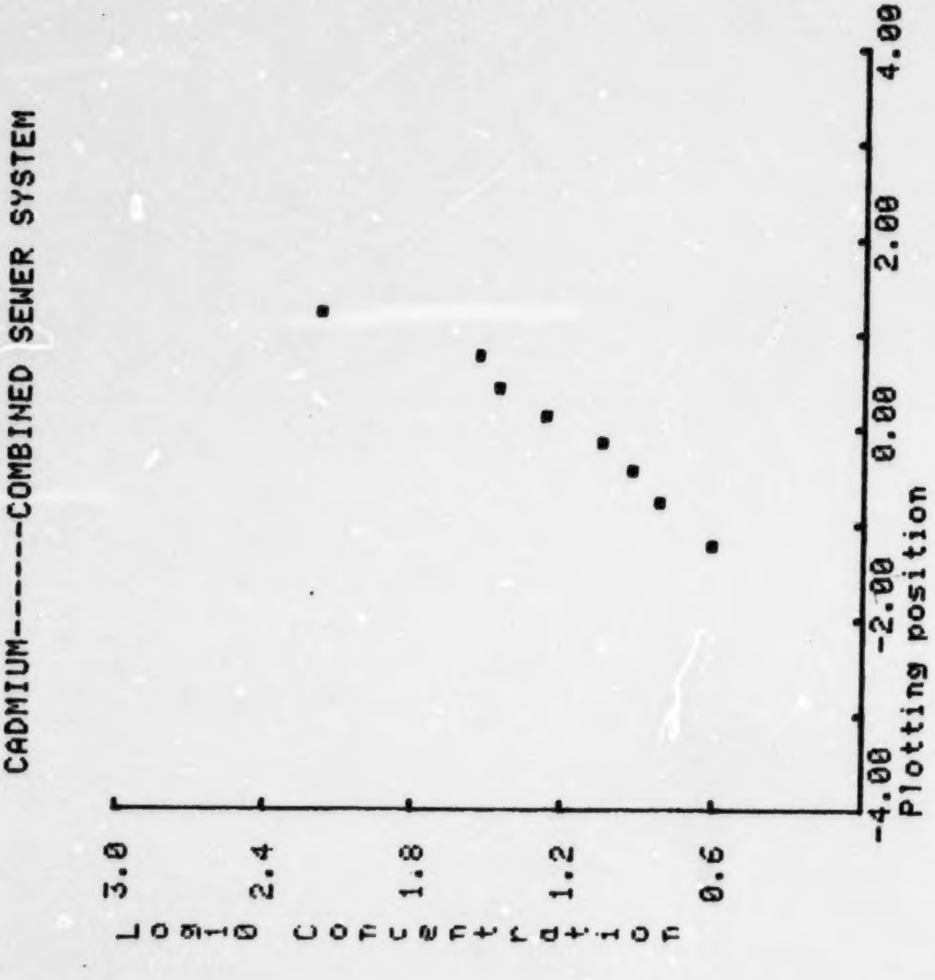

CADMIUM------COMBINED SEWER SYSTEM

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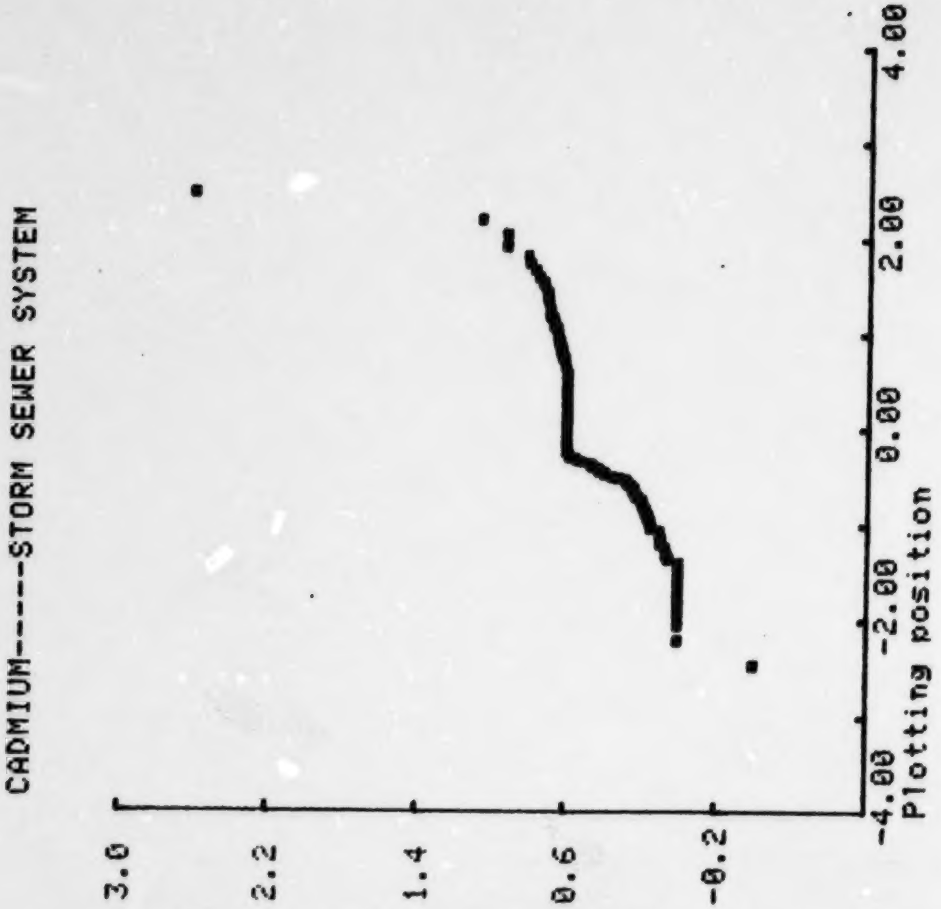

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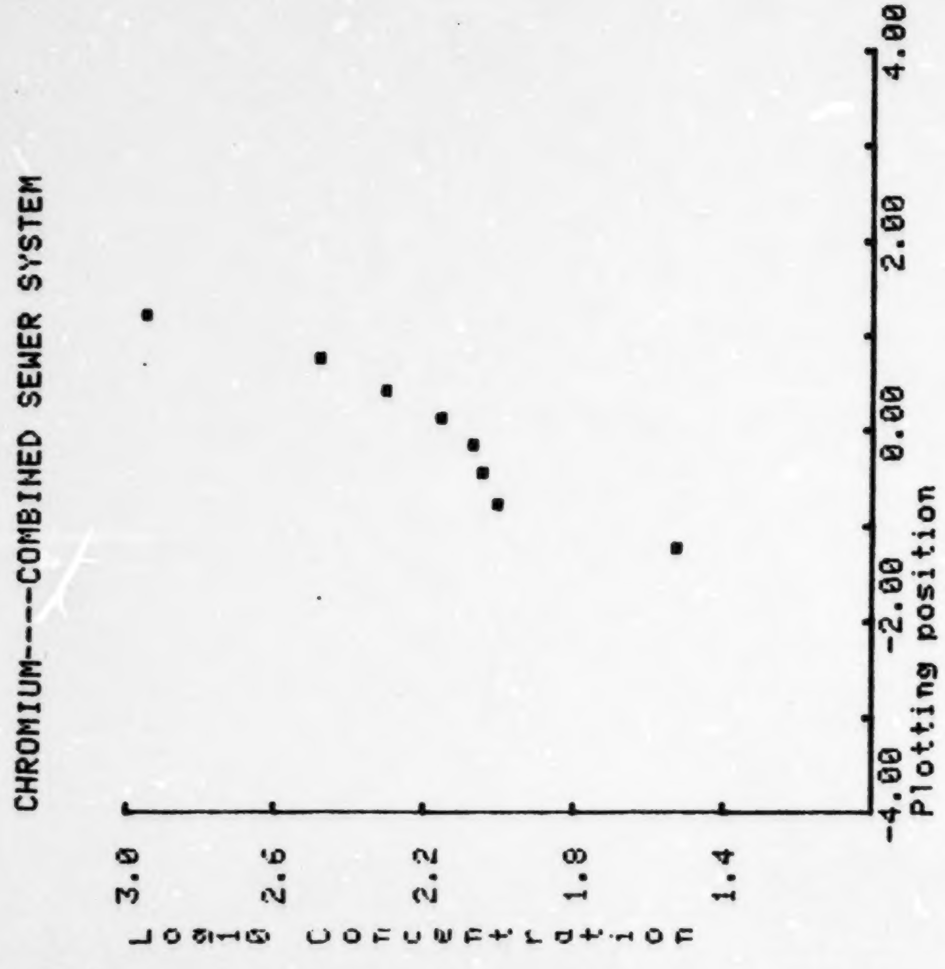

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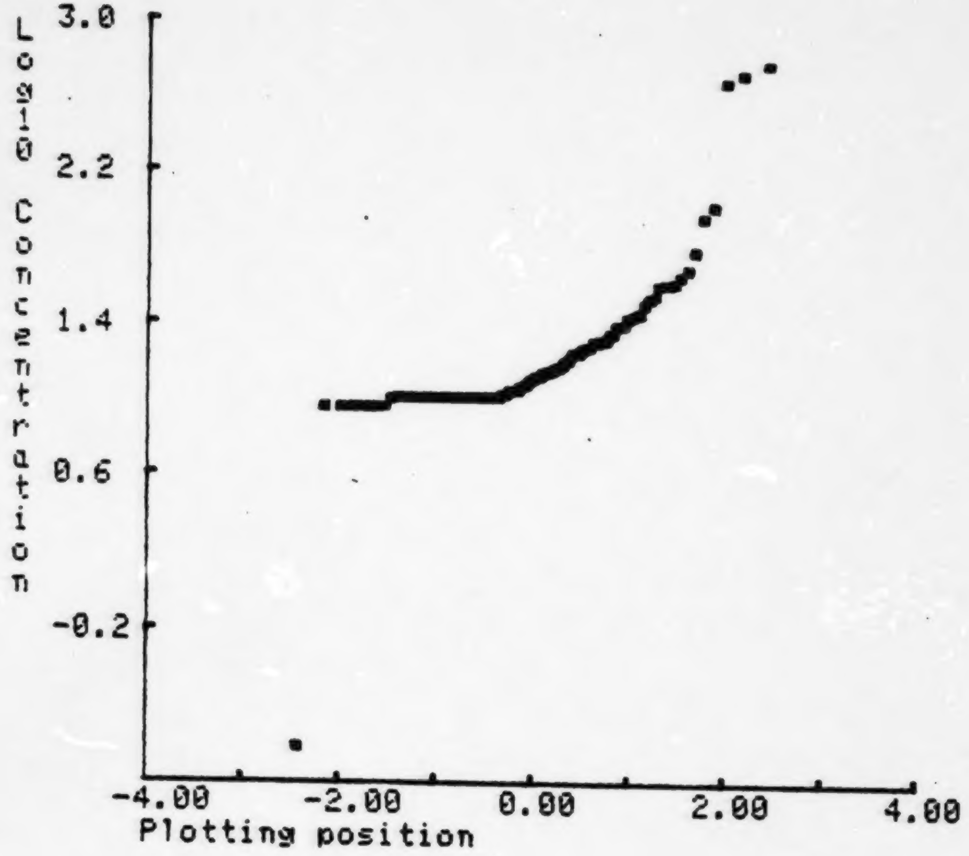
CHROMIUM-STORM SEWER SYSTEM



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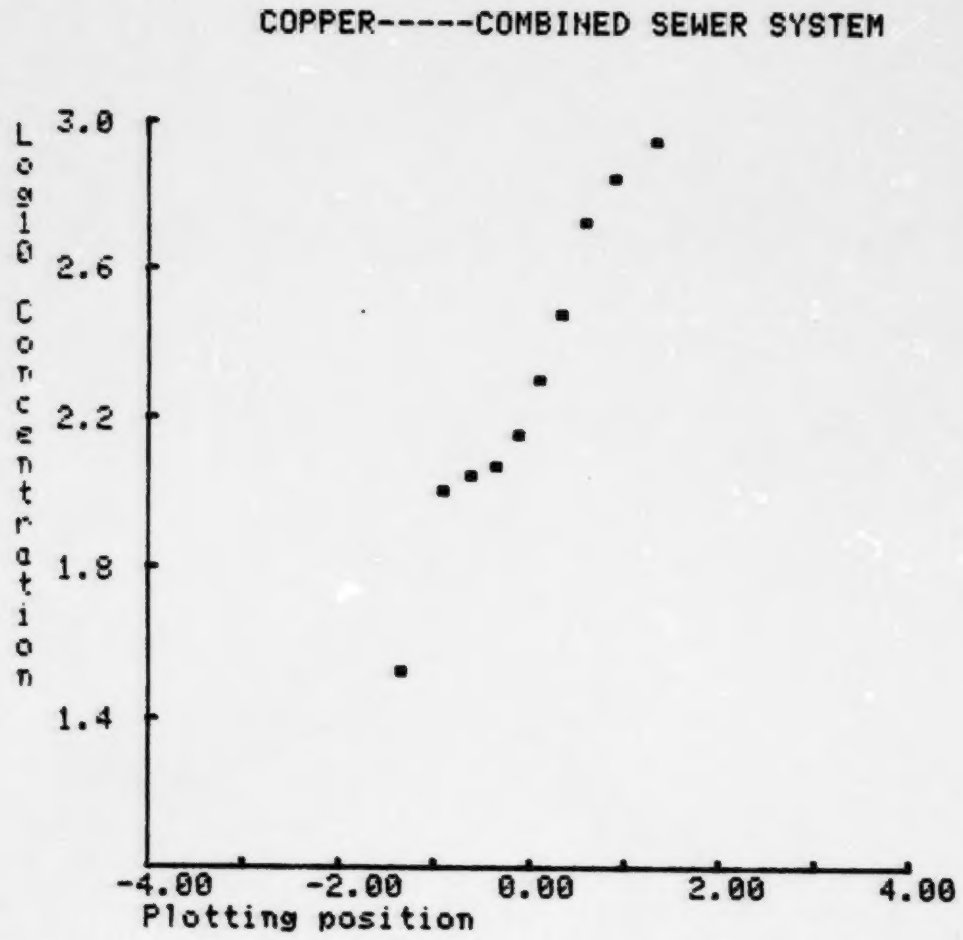

COPPER-----COMBINED SEWER SYSTEM

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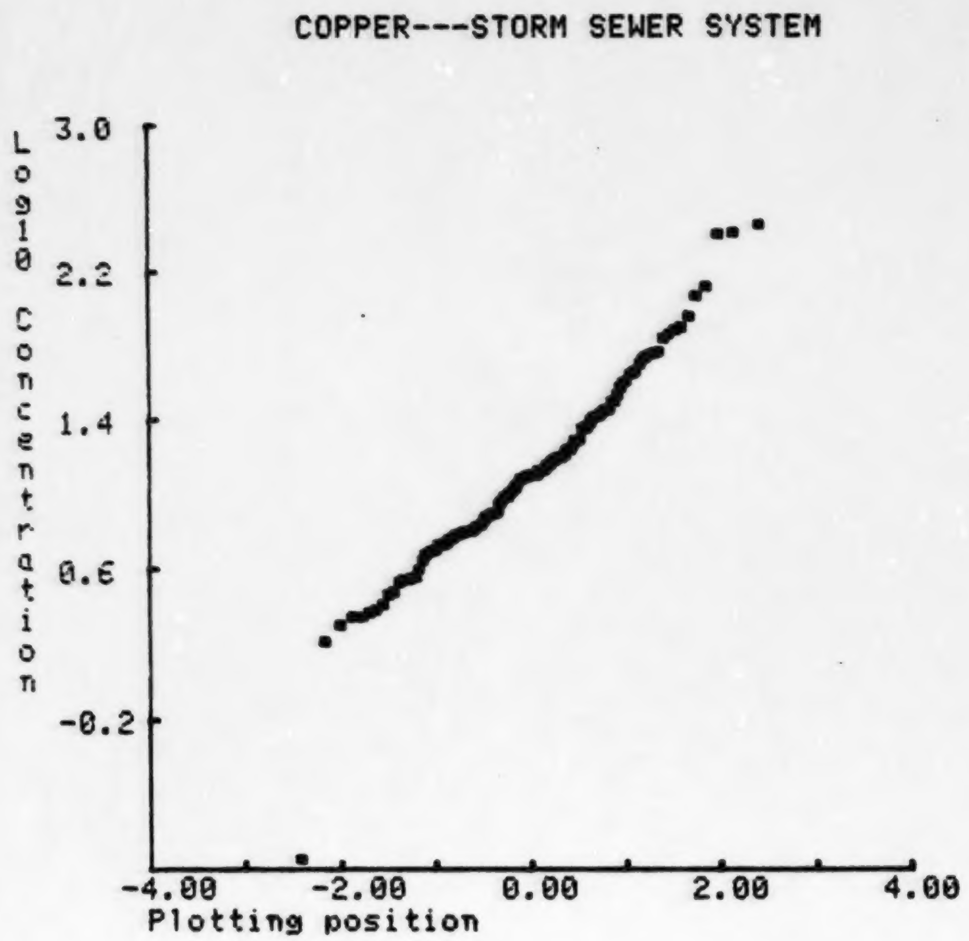

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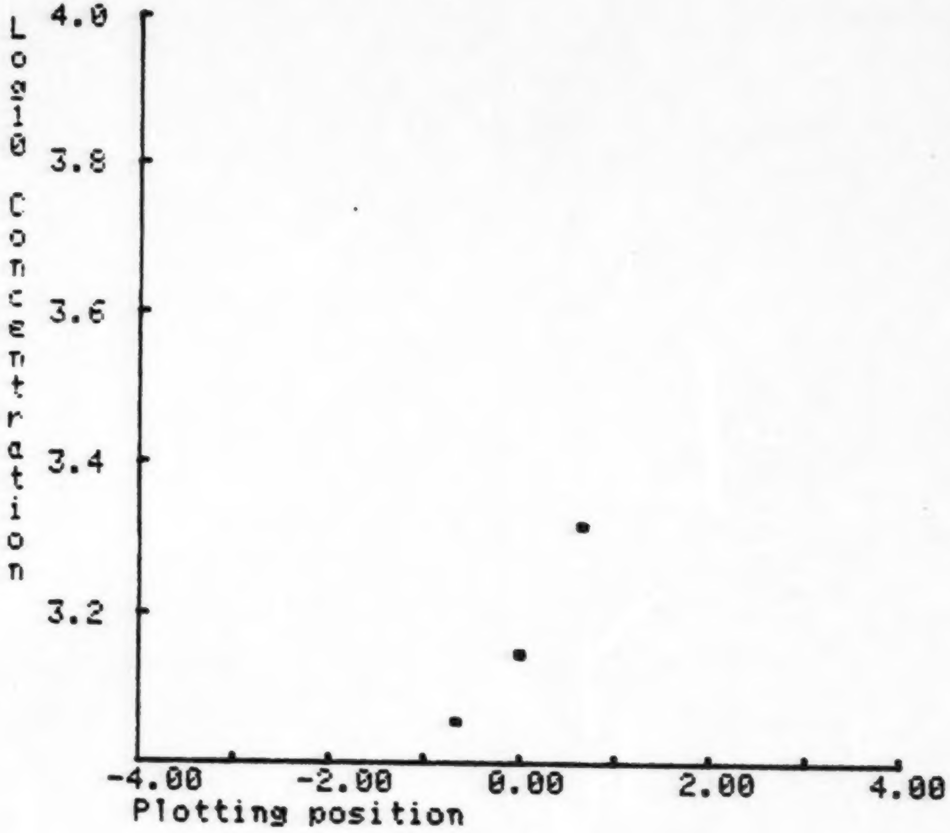
IRON-----COMBINED SEWER SYSTEM



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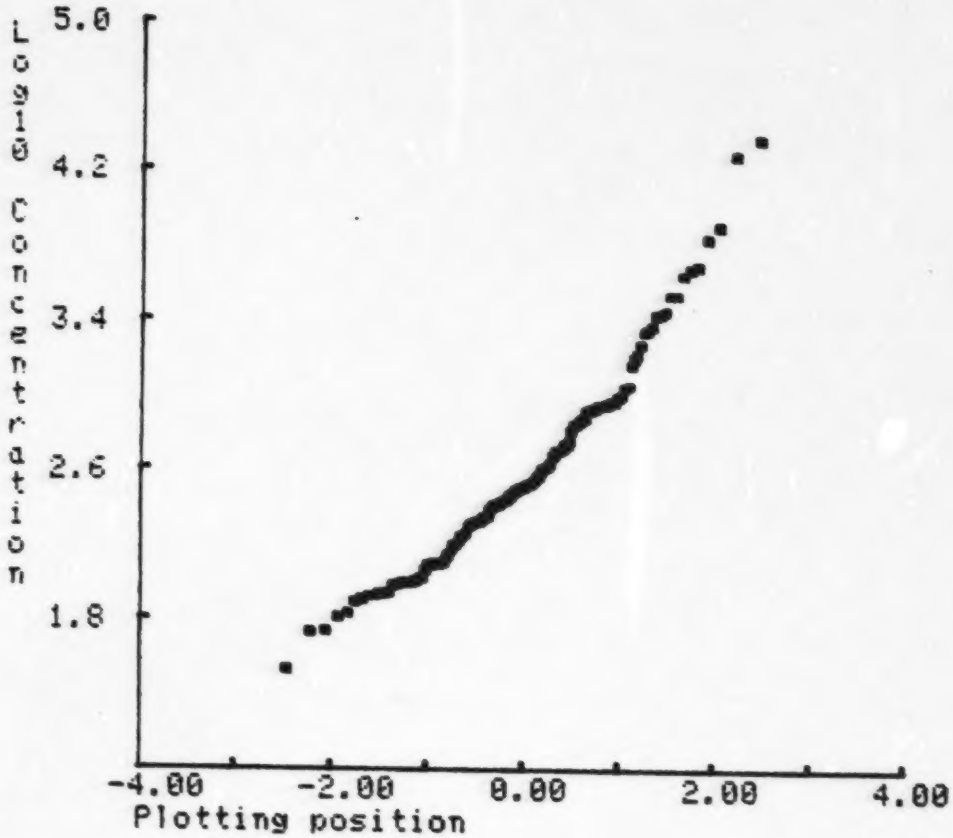
IRON------STORM SEWER SYSTEM



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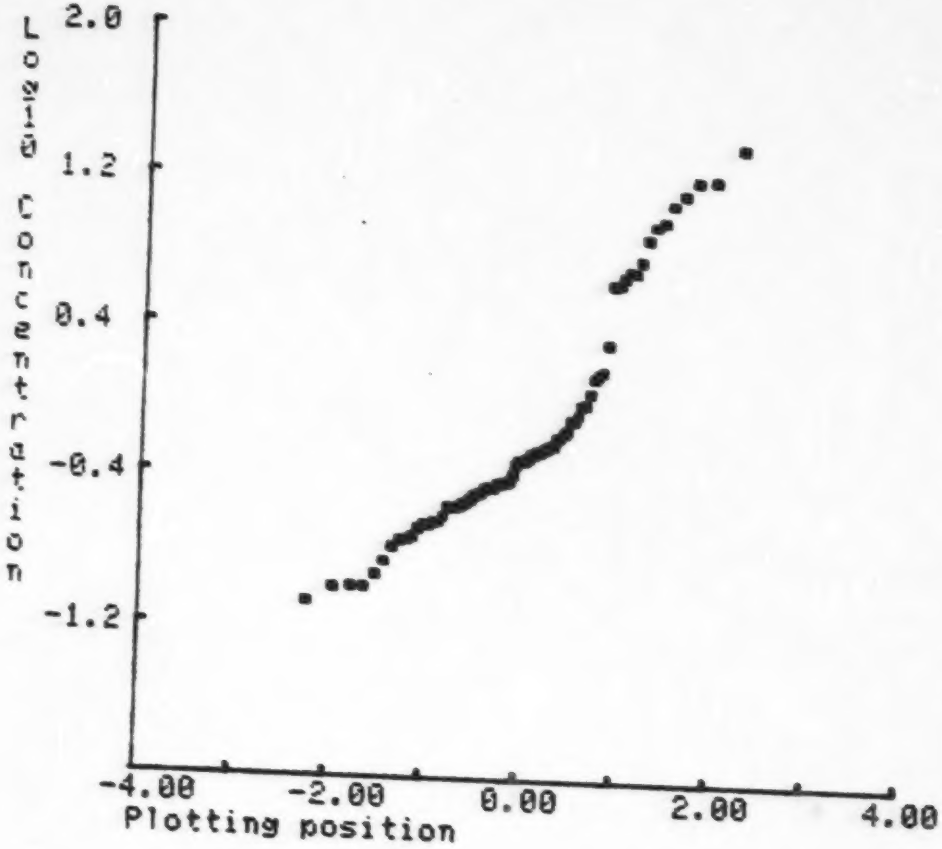
NITRATE-----COMBINED SEWER SYSTEM



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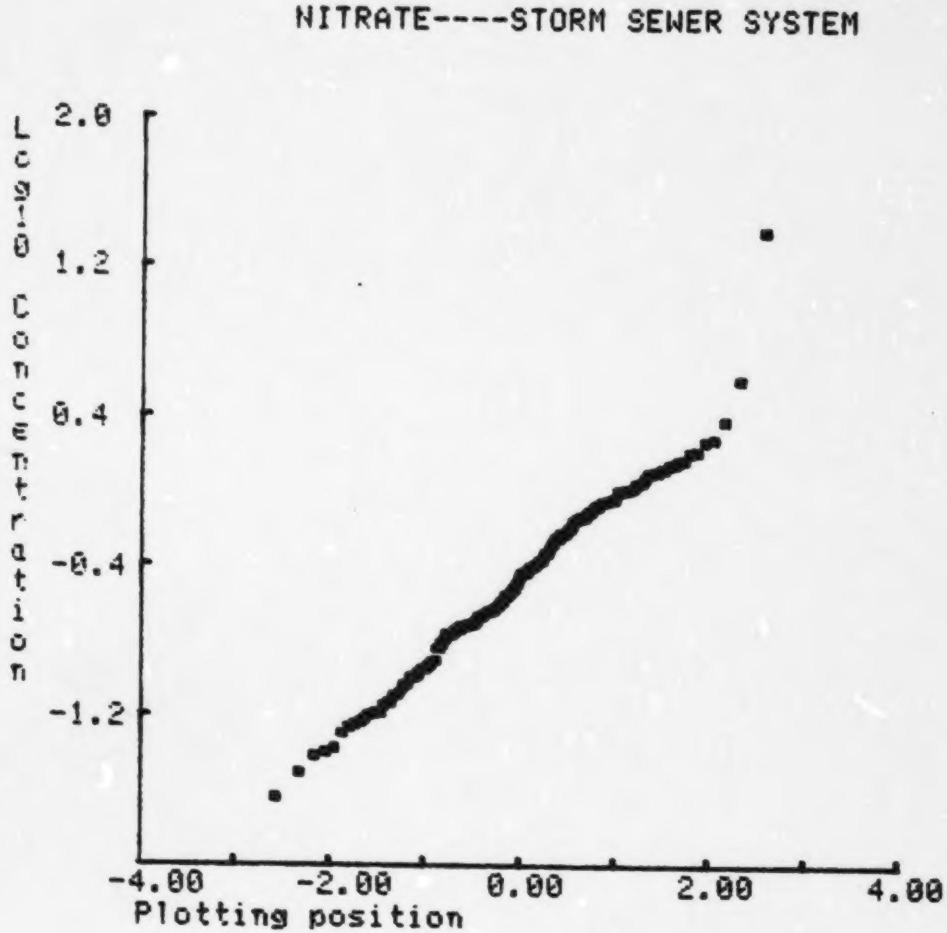

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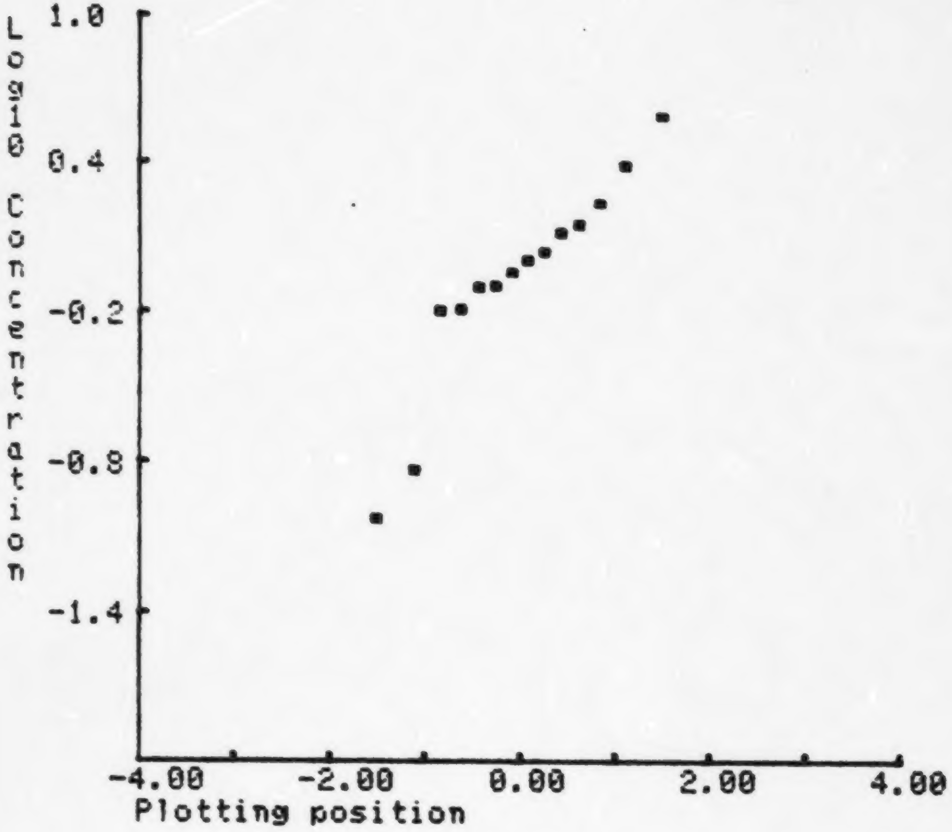
ORTHO-P-----COMBINED STORM SEWER



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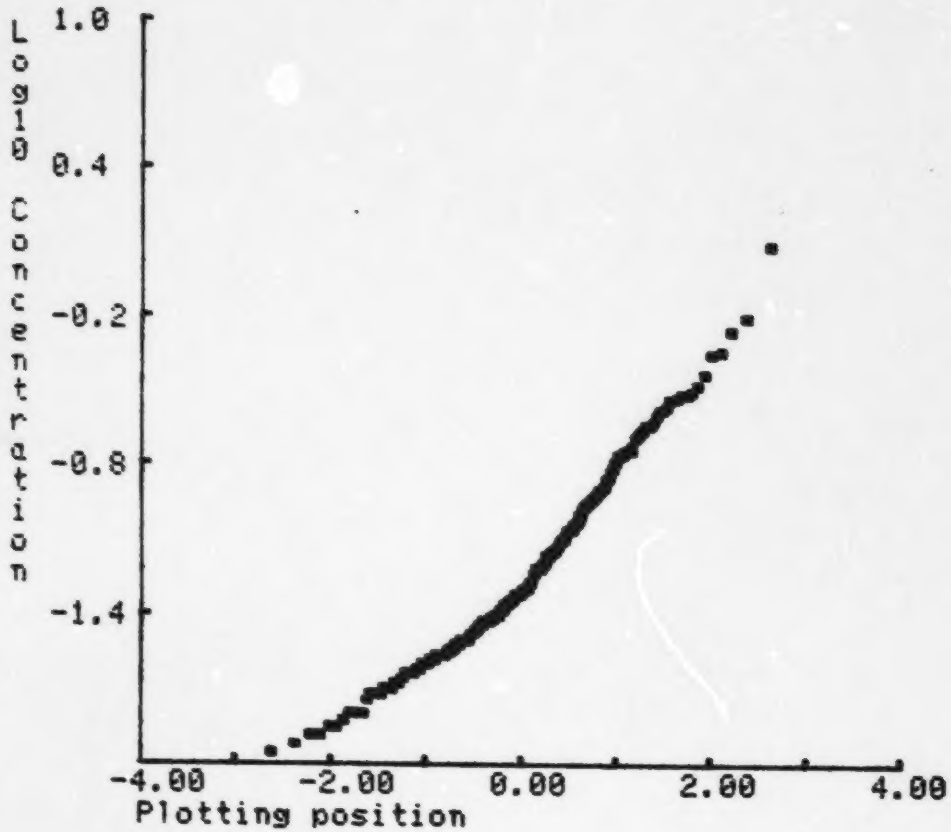
ORTHO-P----STORM SEWER SYSTEM



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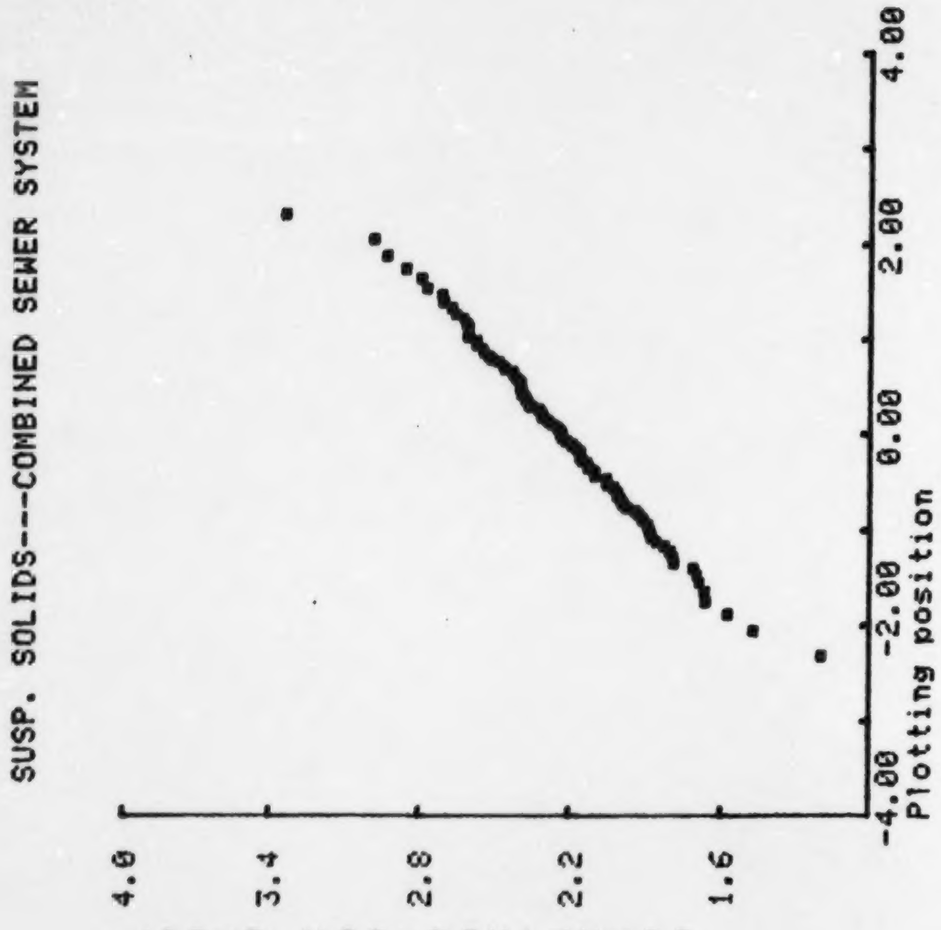

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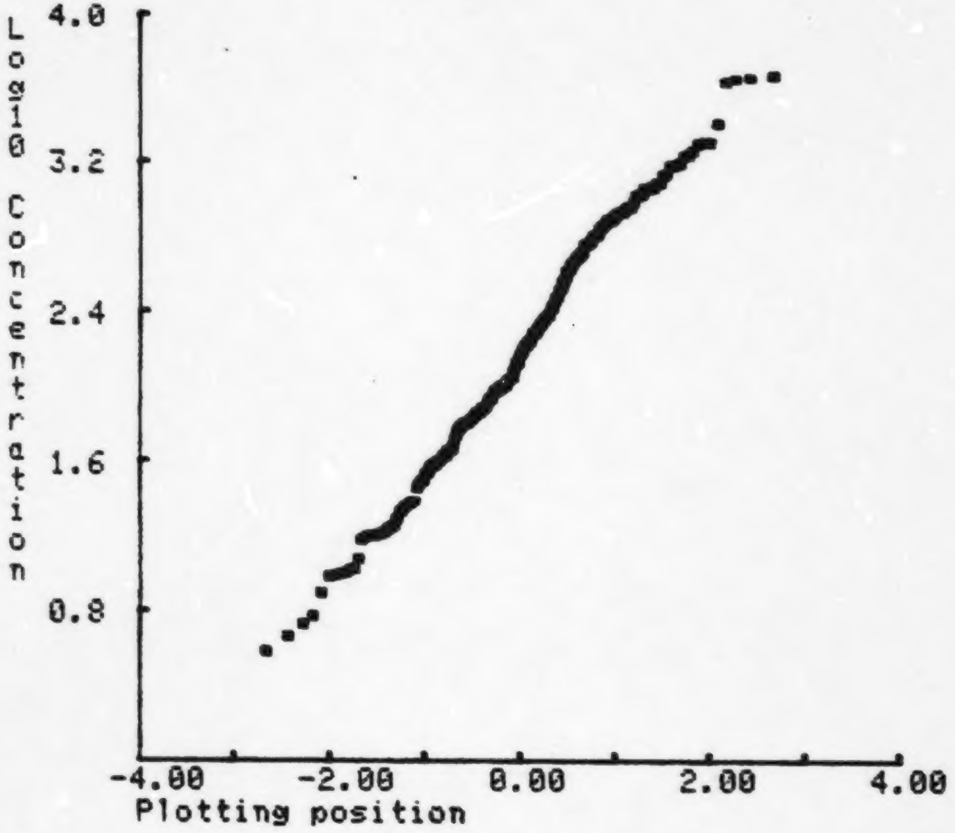
SUSPENDED SOLIDS----STORM SEWER SYSTEM



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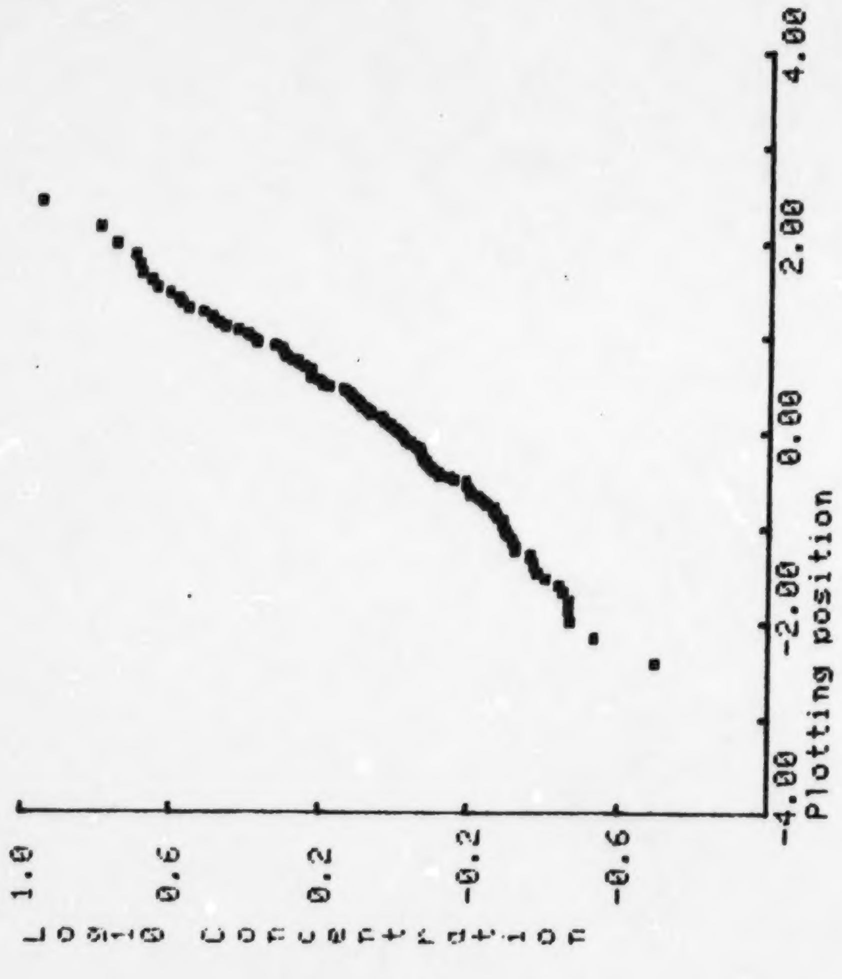
TOTAL N (AS NITROGEN)---STORM SEWER SYSTEM



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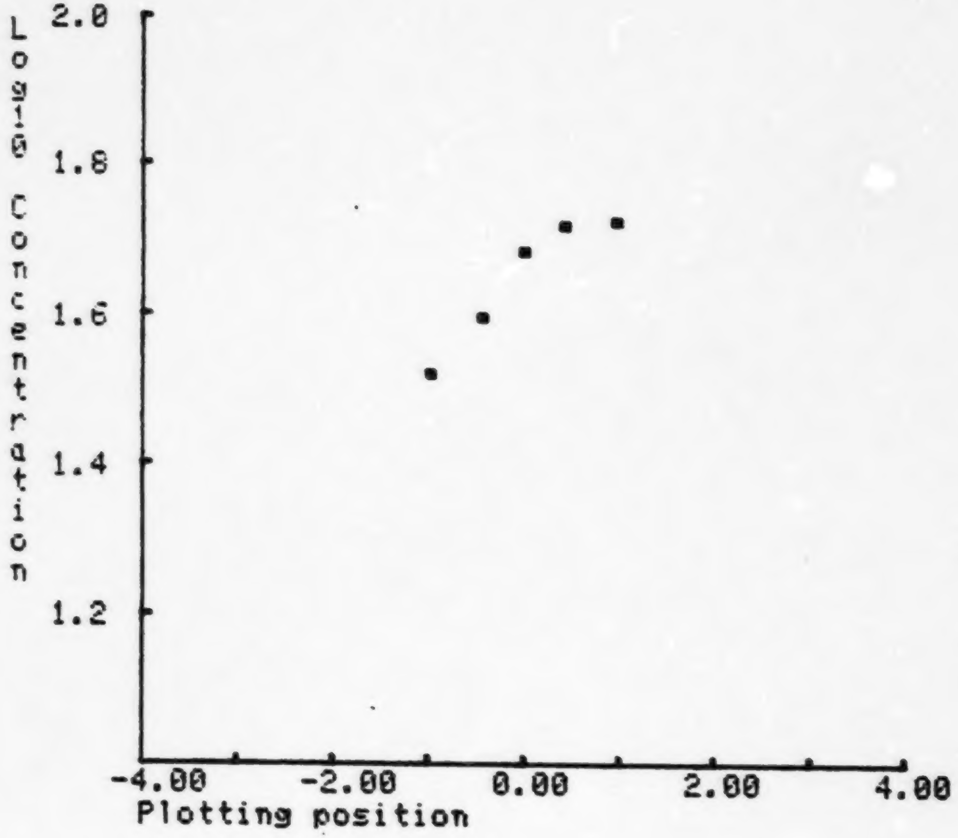
TURBIDITY-----COMBINED SEWER SYSTEM

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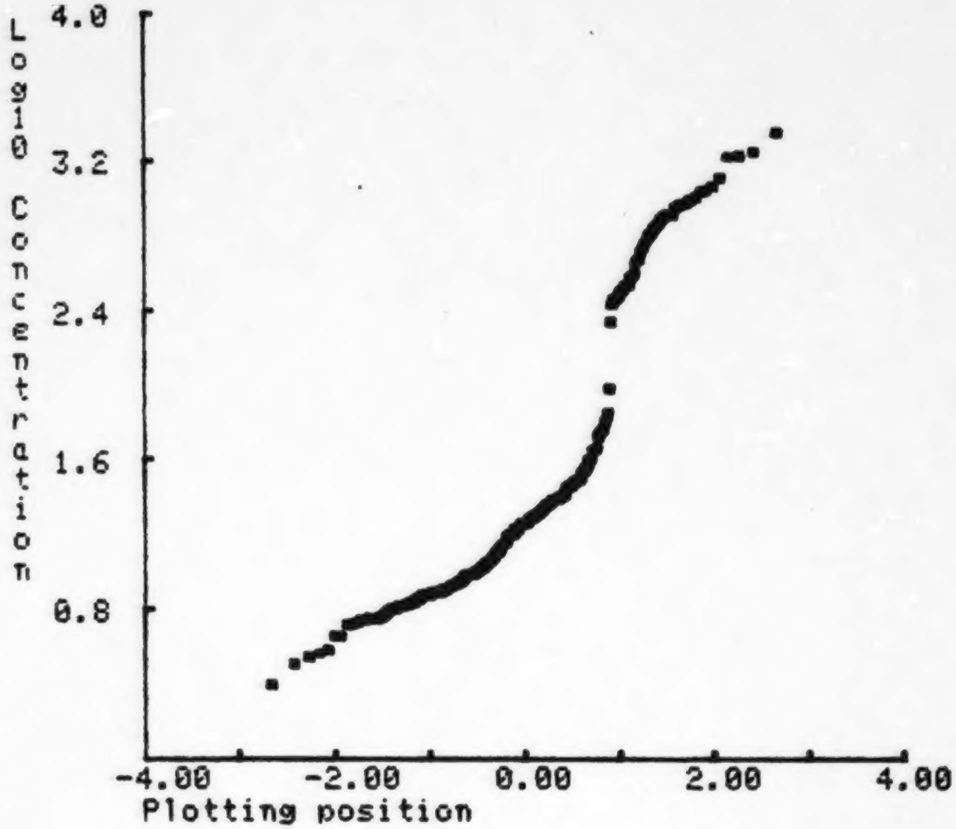
best available copy.

TURBIDITY---STORM SEWER SYSTEM



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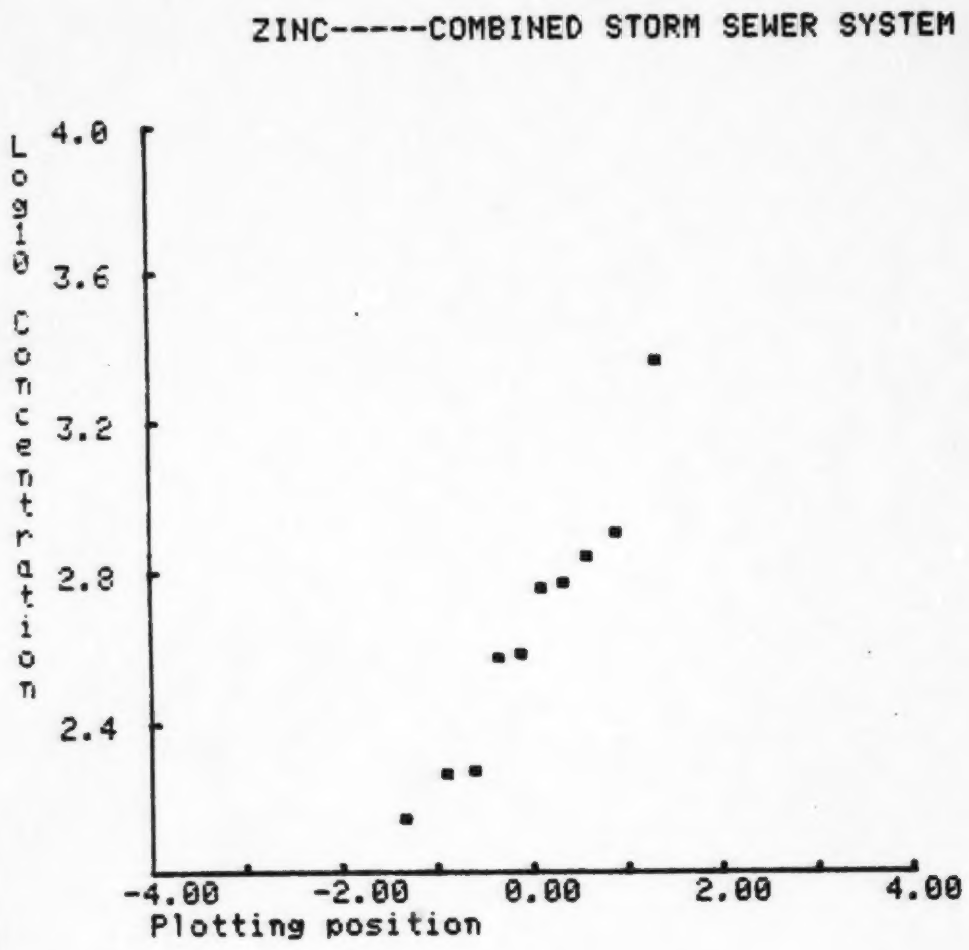

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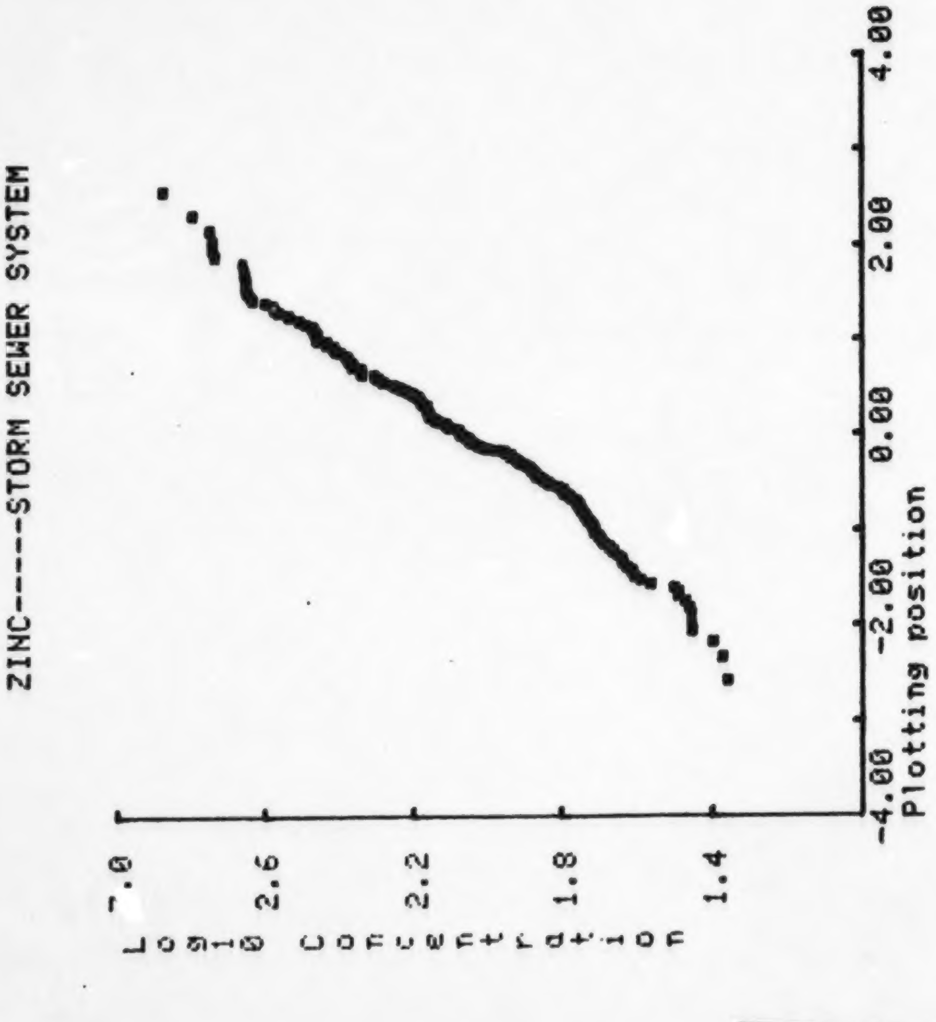
ZINC-----STORM SEWER SYSTEM

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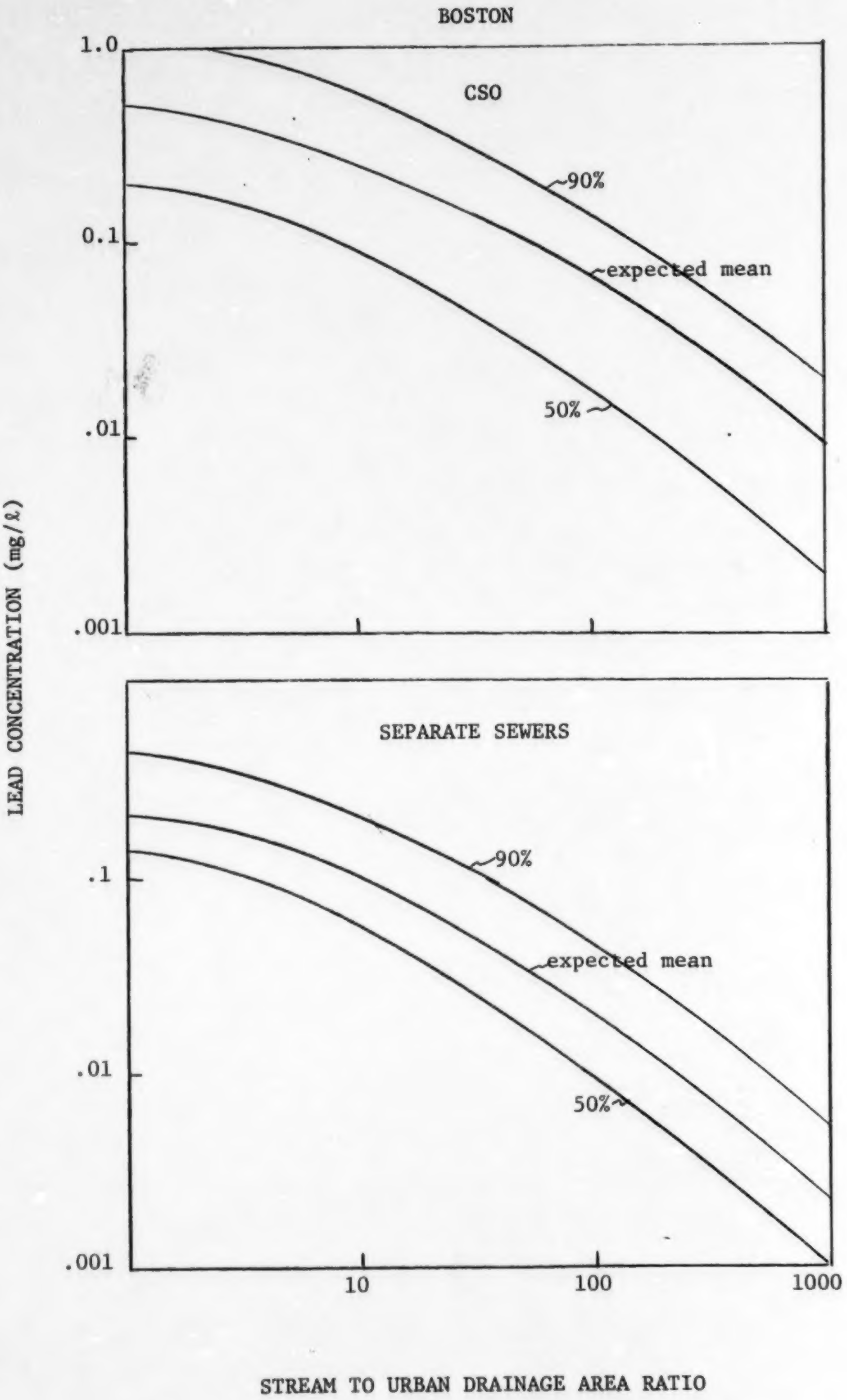
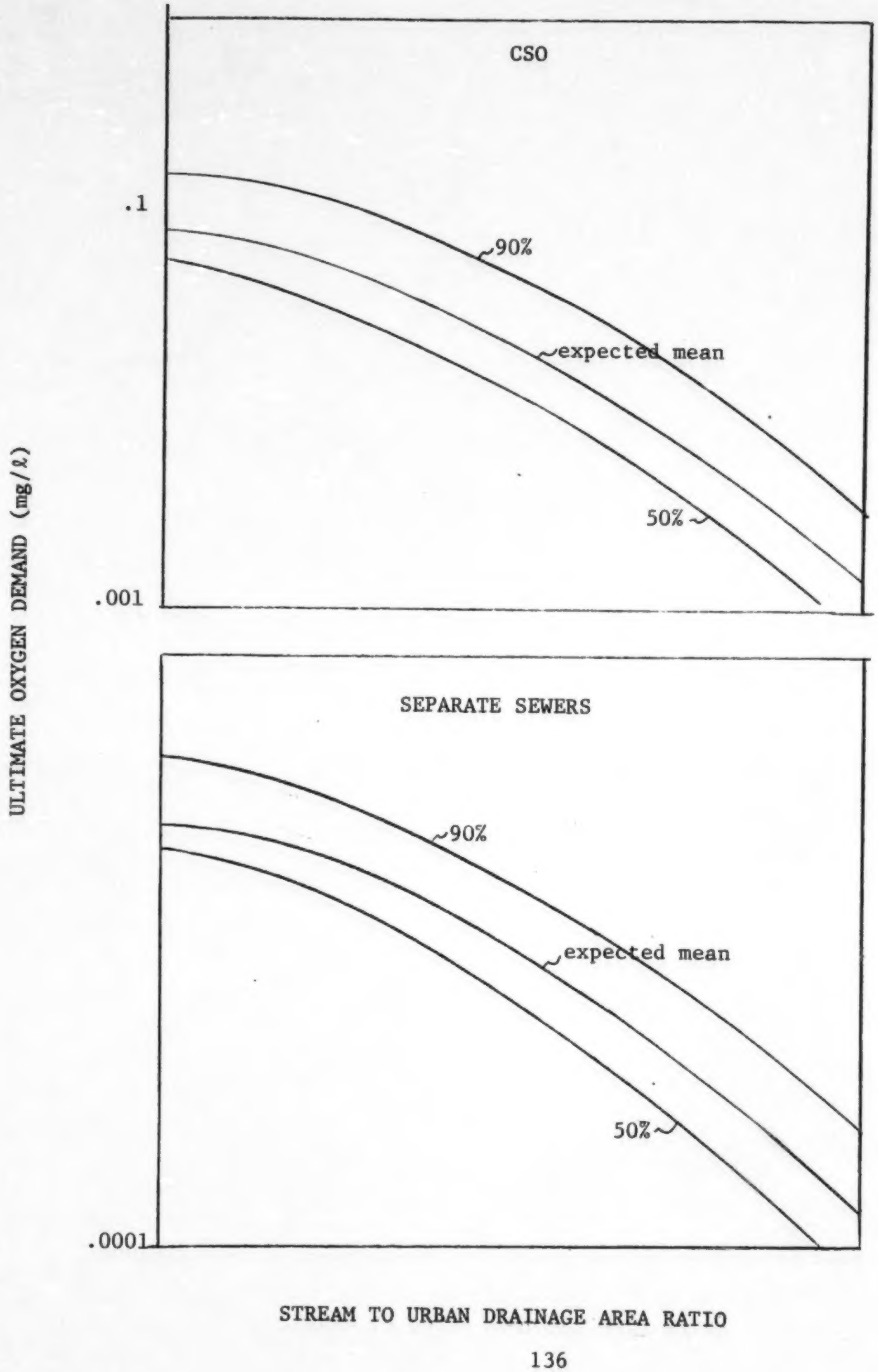
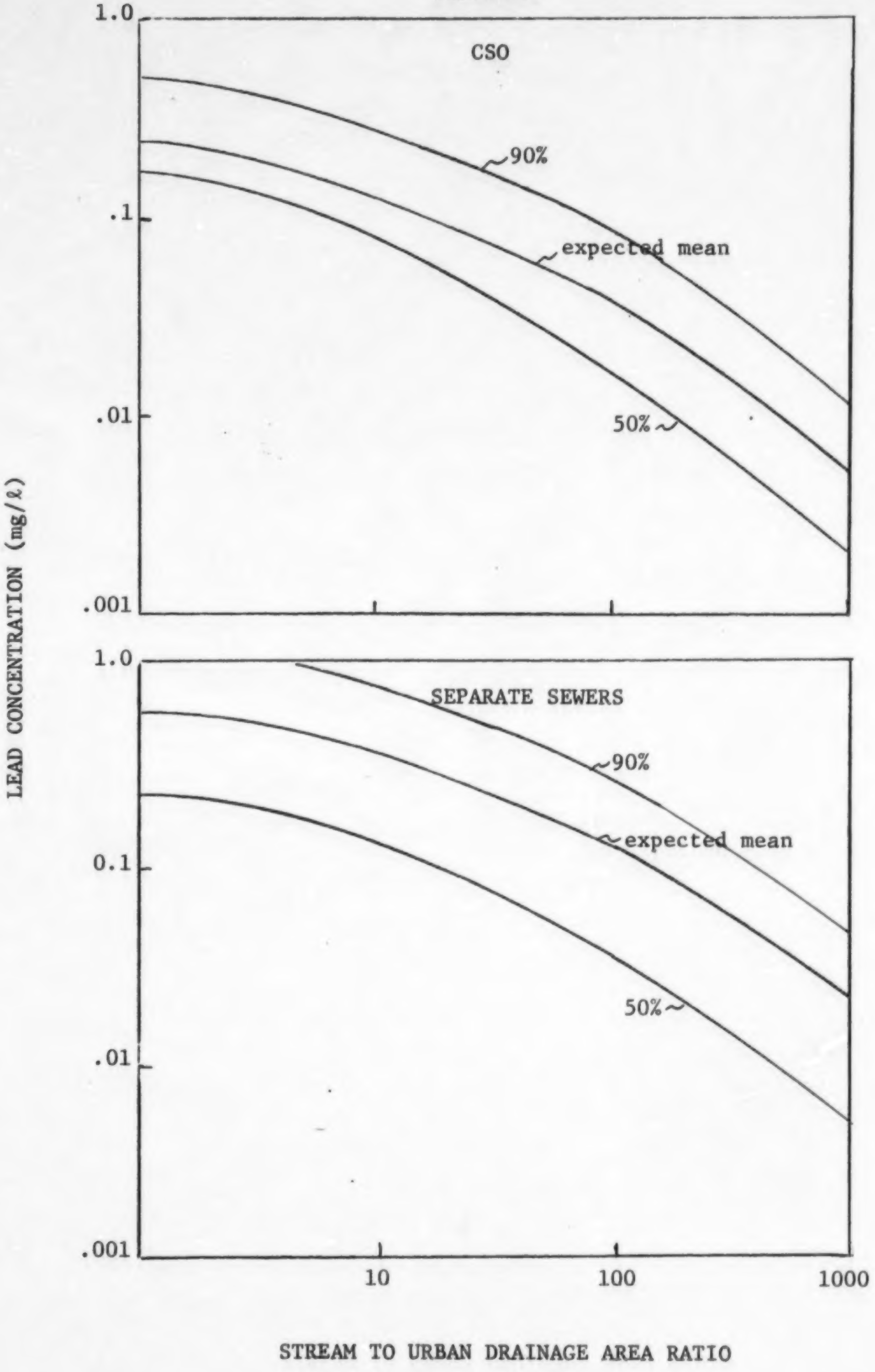

BOSTON
CSO
1.0
0.1
.01
.001
~90%
~expected mean
50%~
SEPARATE SEWERS
.1
.01
.001
~90%
~expected mean
50%~
10
100
1000
LEAD CONCENTRATION (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO

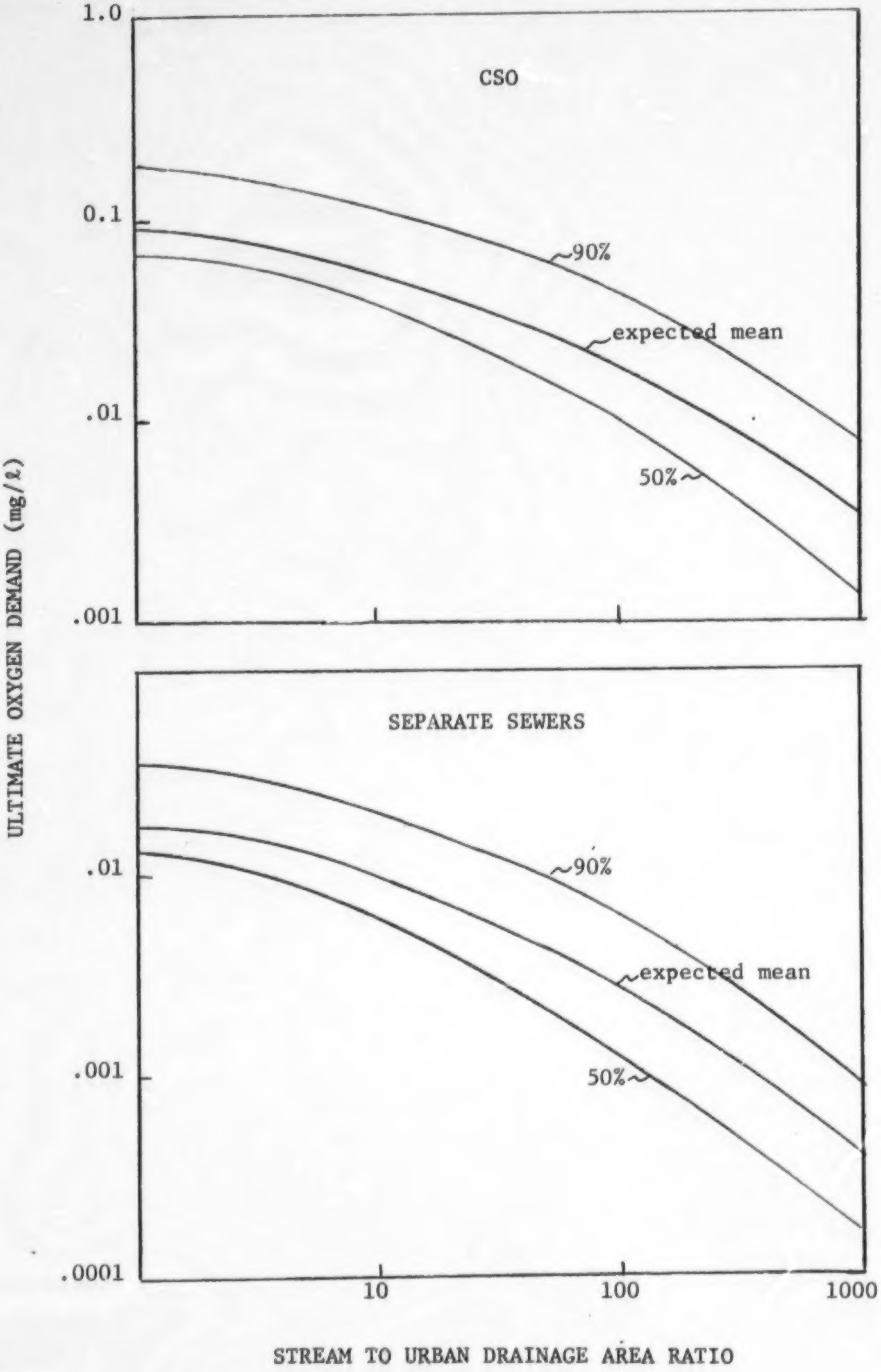
BOSTON

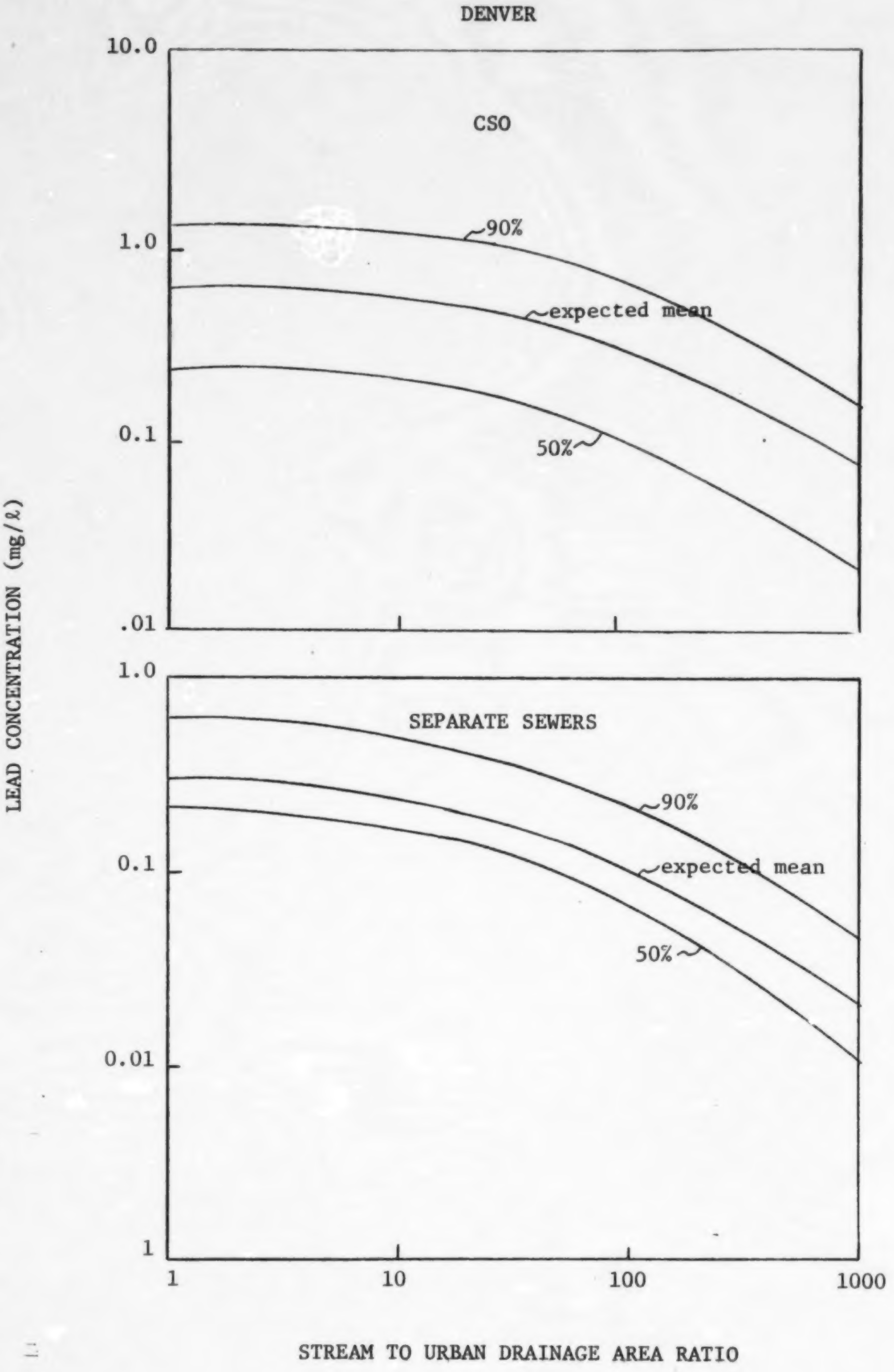

CSO
.1
~90%
~expected mean
50%~
.001
SEPARATE SEWERS
~90%
~expected mean
50%~
.0001
ULTIMATE OXYGEN DEMAND (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO

COLUMBIA

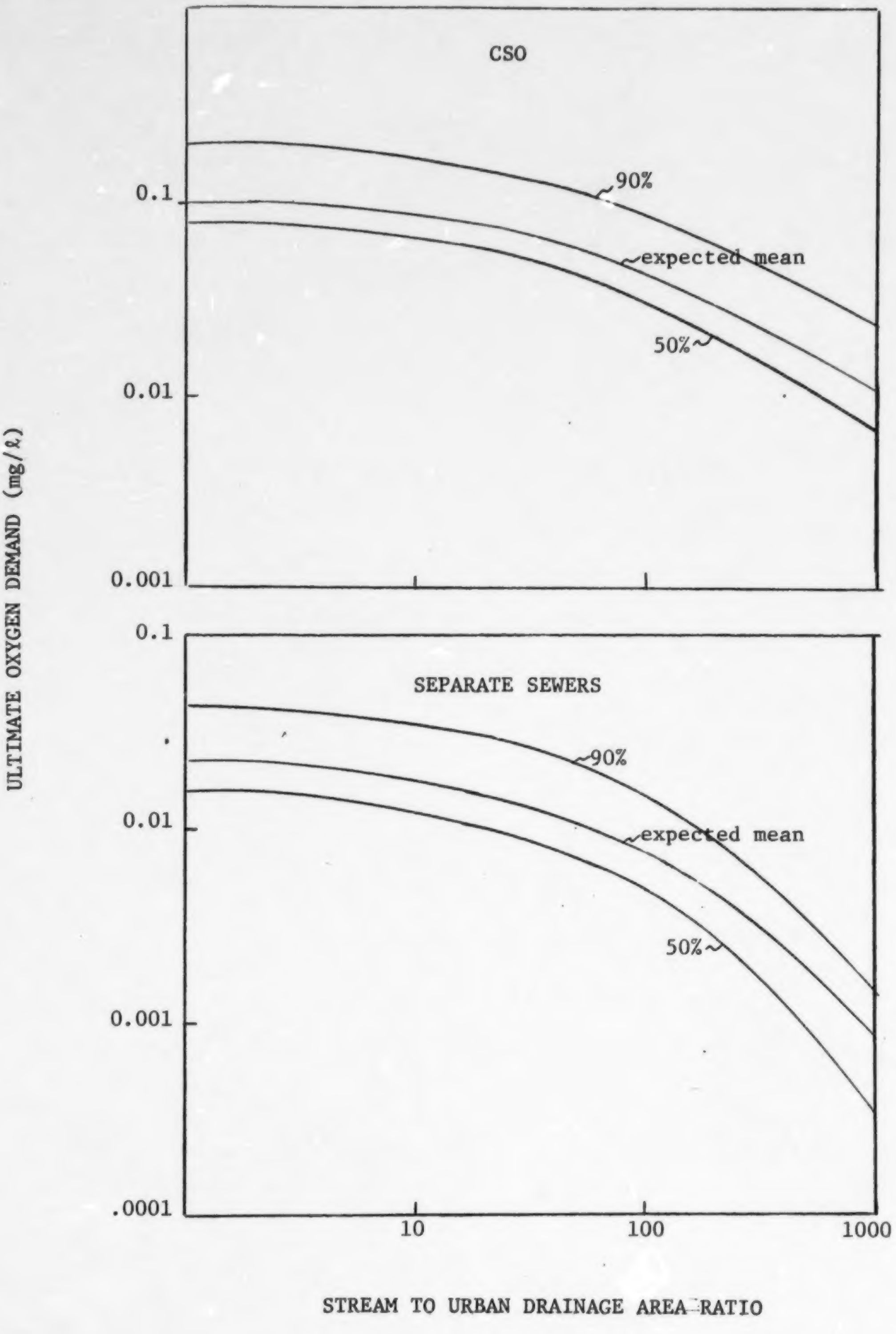

CSO
90%
expected mean
50%
SEPARATE SEWERS
90%
expected mean
50%
1.0
.1
.01
.001
1.0
0.1
.01
.001
10
100
1000
LEAD CONCENTRATION (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO

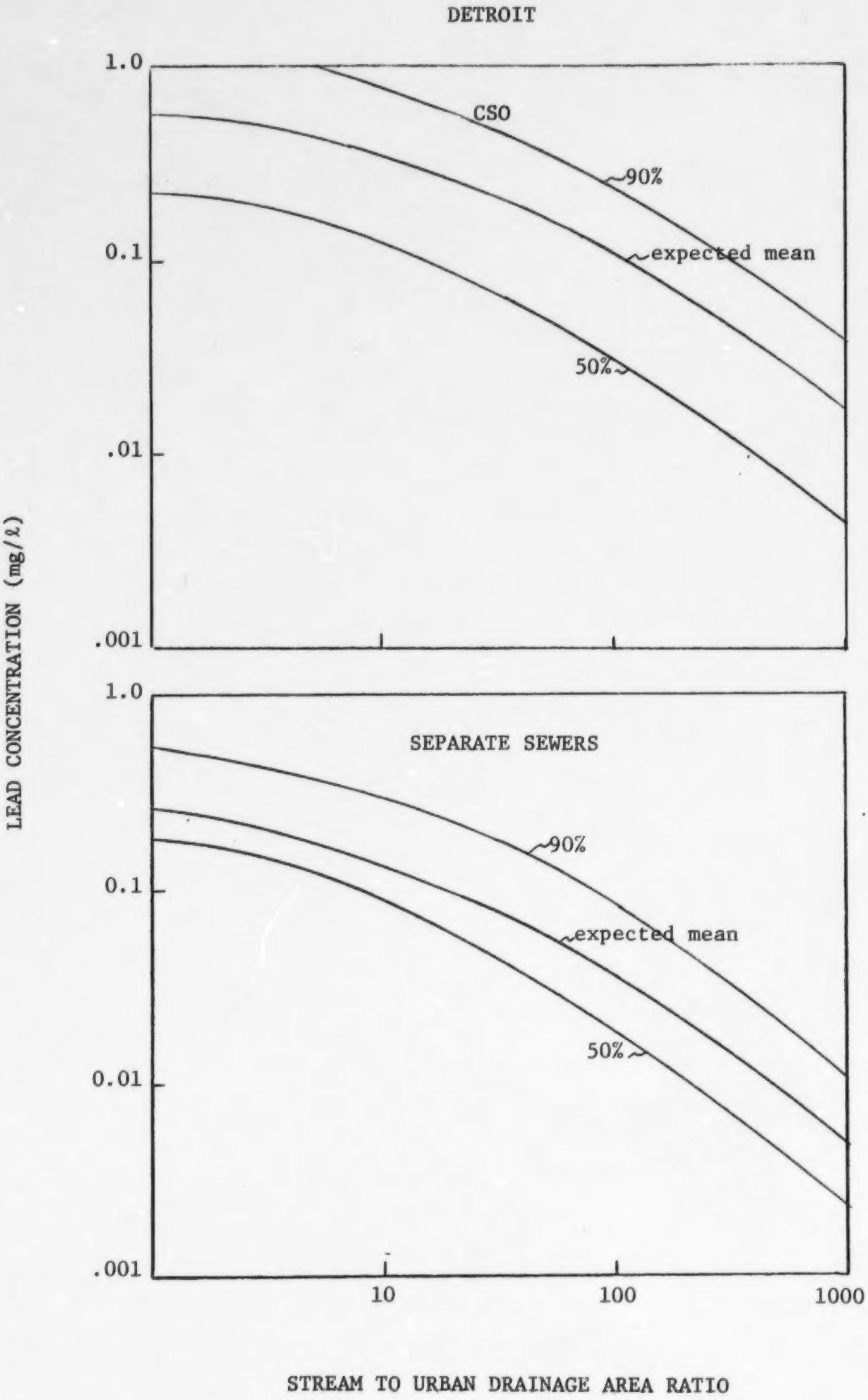
COLUMBIA


CSO
1.0
0.1
.01
.001
~90%
expected mean
50%
SEPARATE SEWERS
.01
.001
.0001
~90%
expected mean
50%
10
100
1000
ULTIMATE OXYGEN DEMAND (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO

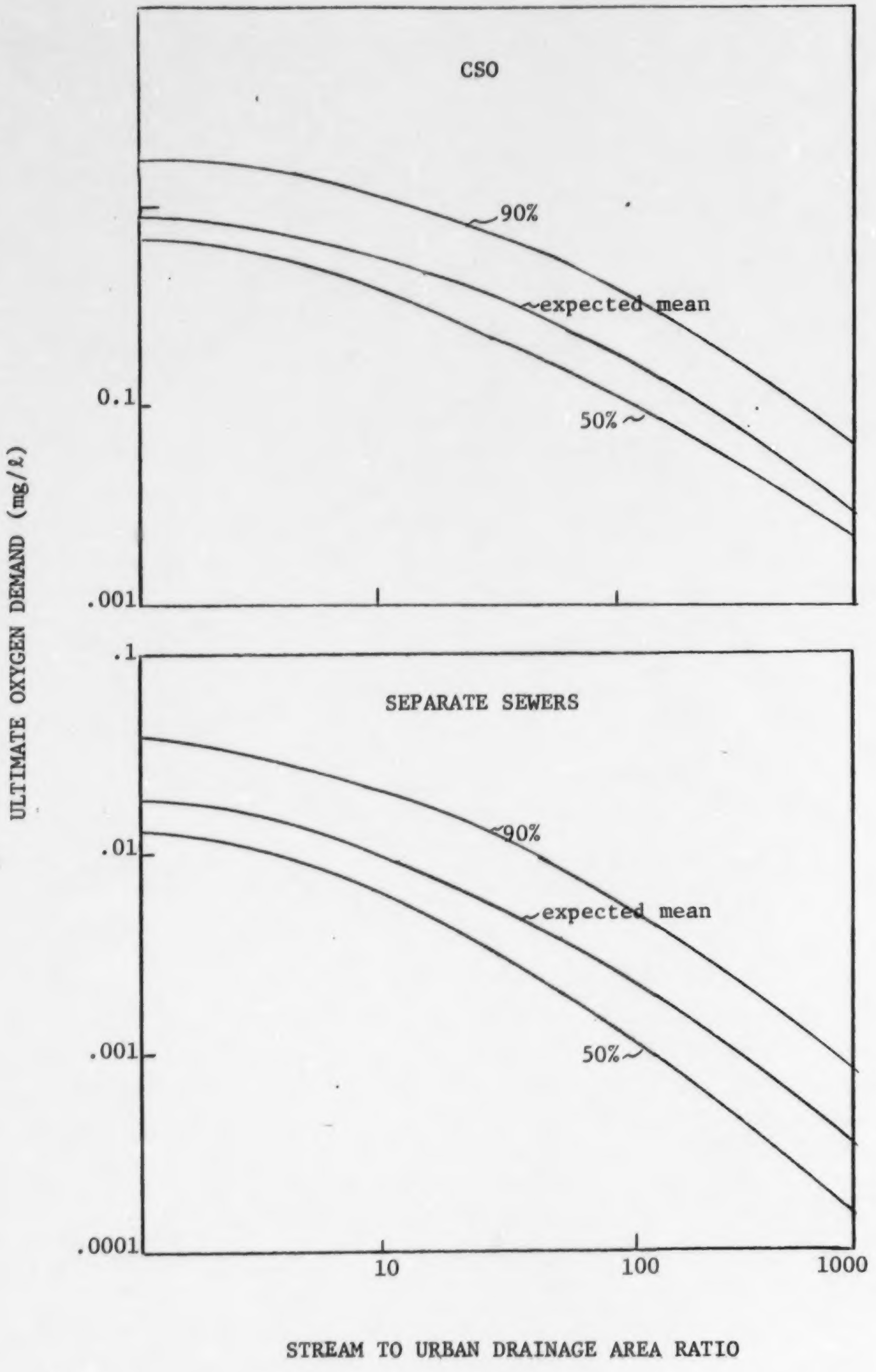

DENVER
10.0
1.0
0.1
.01
CSO
90%
expected mean
50%
1.0
0.1
0.01
1
SEPARATE SEWERS
90%
expected mean
50%
1
10
100
1000
LEAD CONCENTRATION (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO

DENVER

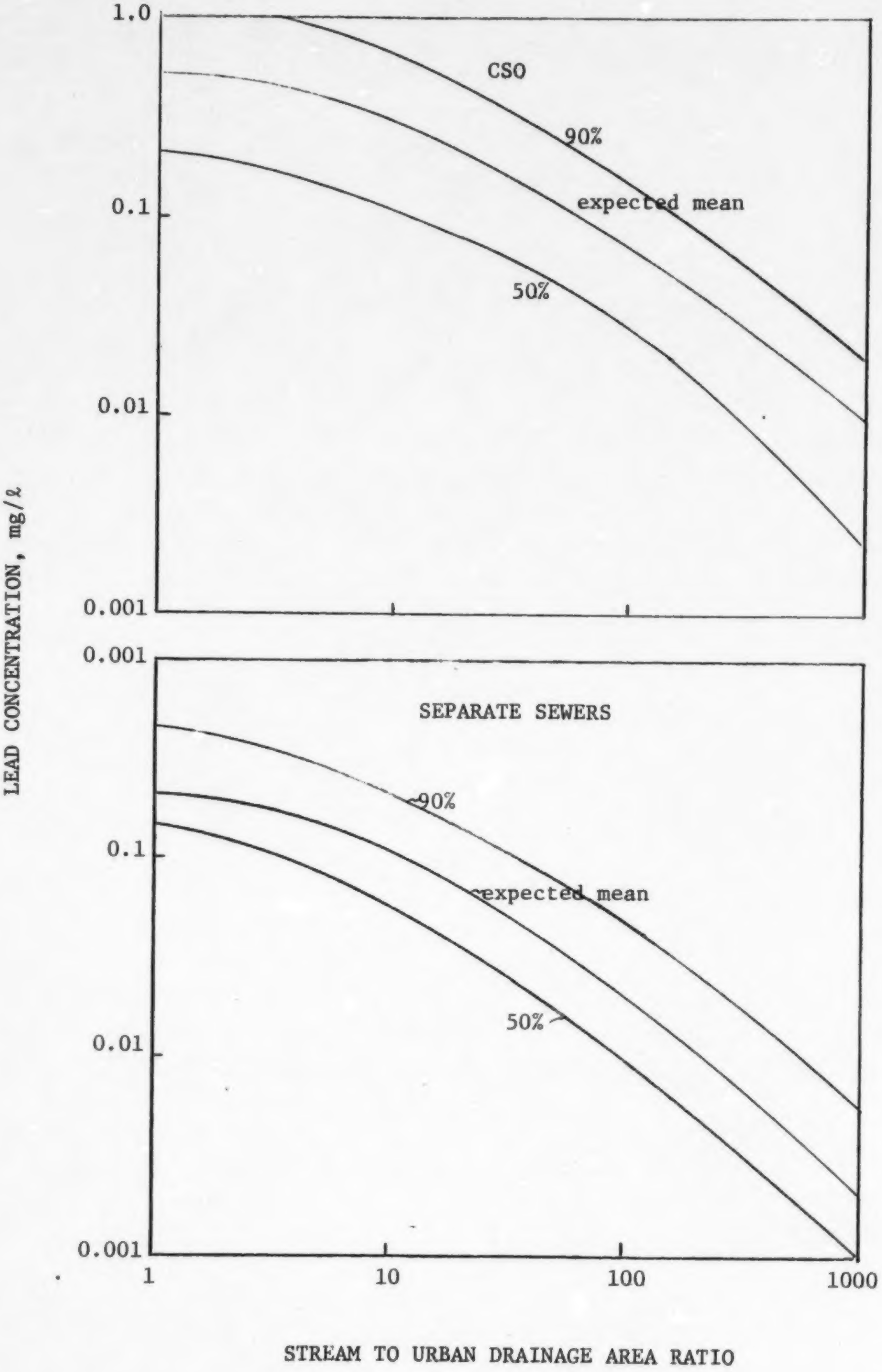

CSO
90%
expected mean
50%
SEPARATE SEWERS
90%
expected mean
50%
0.1
0.01
0.001
0.1
0.01
0.001
.0001
10
100
1000
ULTIMATE OXYGEN DEMAND (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO


DETROIT
CSO
90%
expected mean
50%
SEPARATE SEWERS
90%
expected mean
50%
1.0
0.1
.01
.001
1.0
0.1
0.01
.001
10
100
1000
LEAD CONCENTRATION (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO

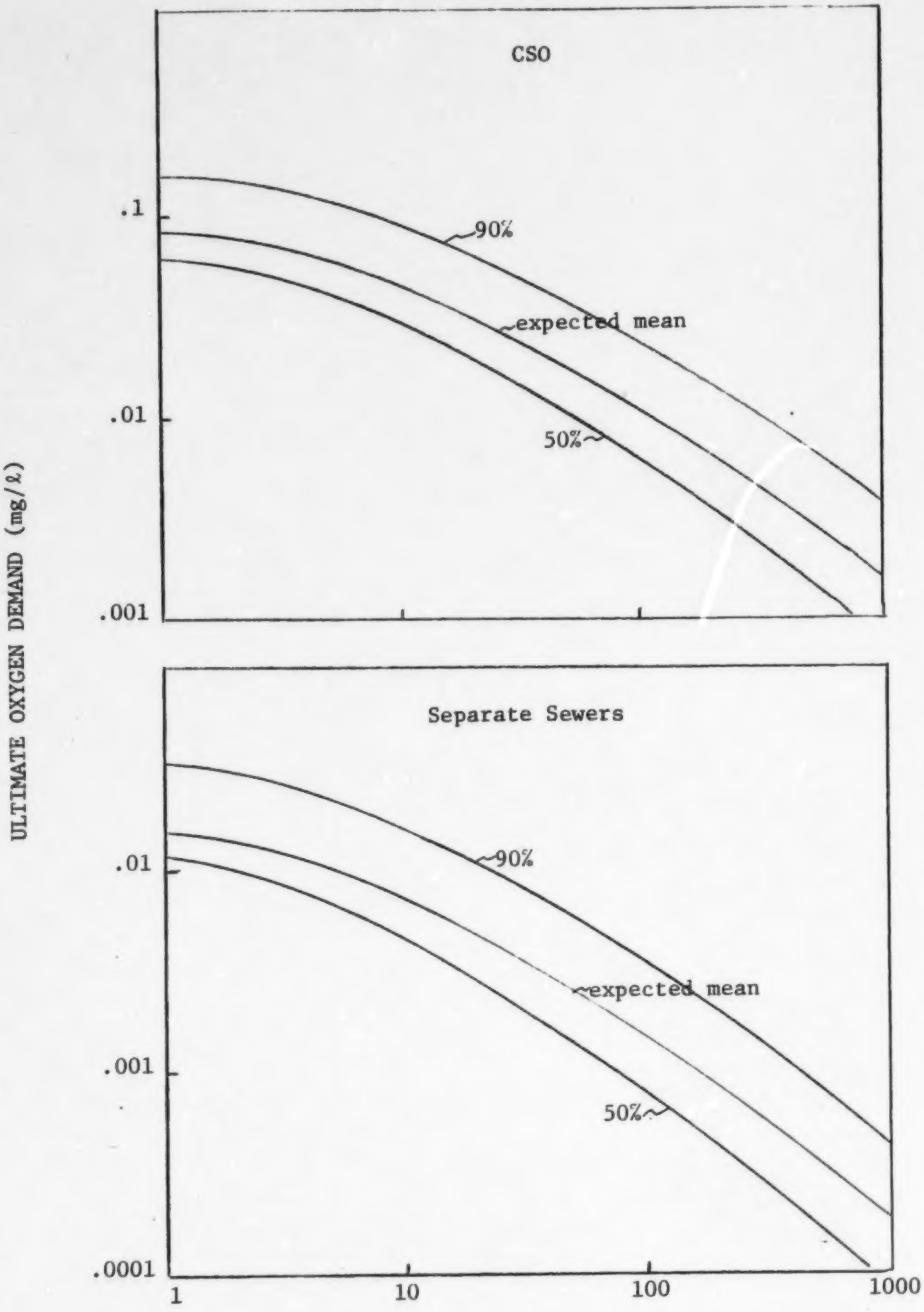
DETROIT


CSO
90%
expected mean
50%
0.1
.001
.1
SEPARATE SEWERS
90%
expected mean
50%
.01
.001
.0001
10
100
1000
ULTIMATE OXYGEN DEMAND (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO

PORTLAND-1 (area a)

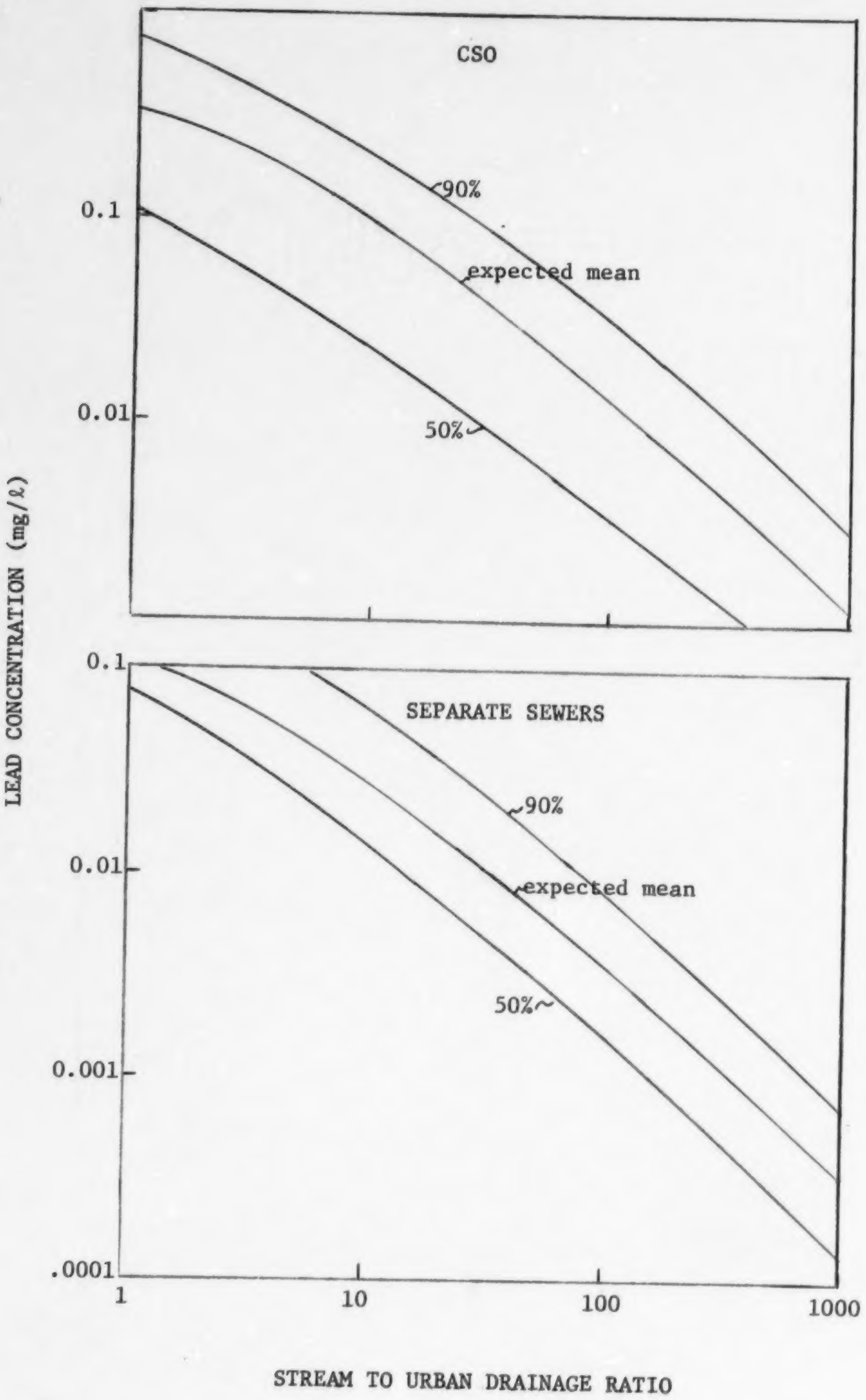


PORTLAND-1 (area a)

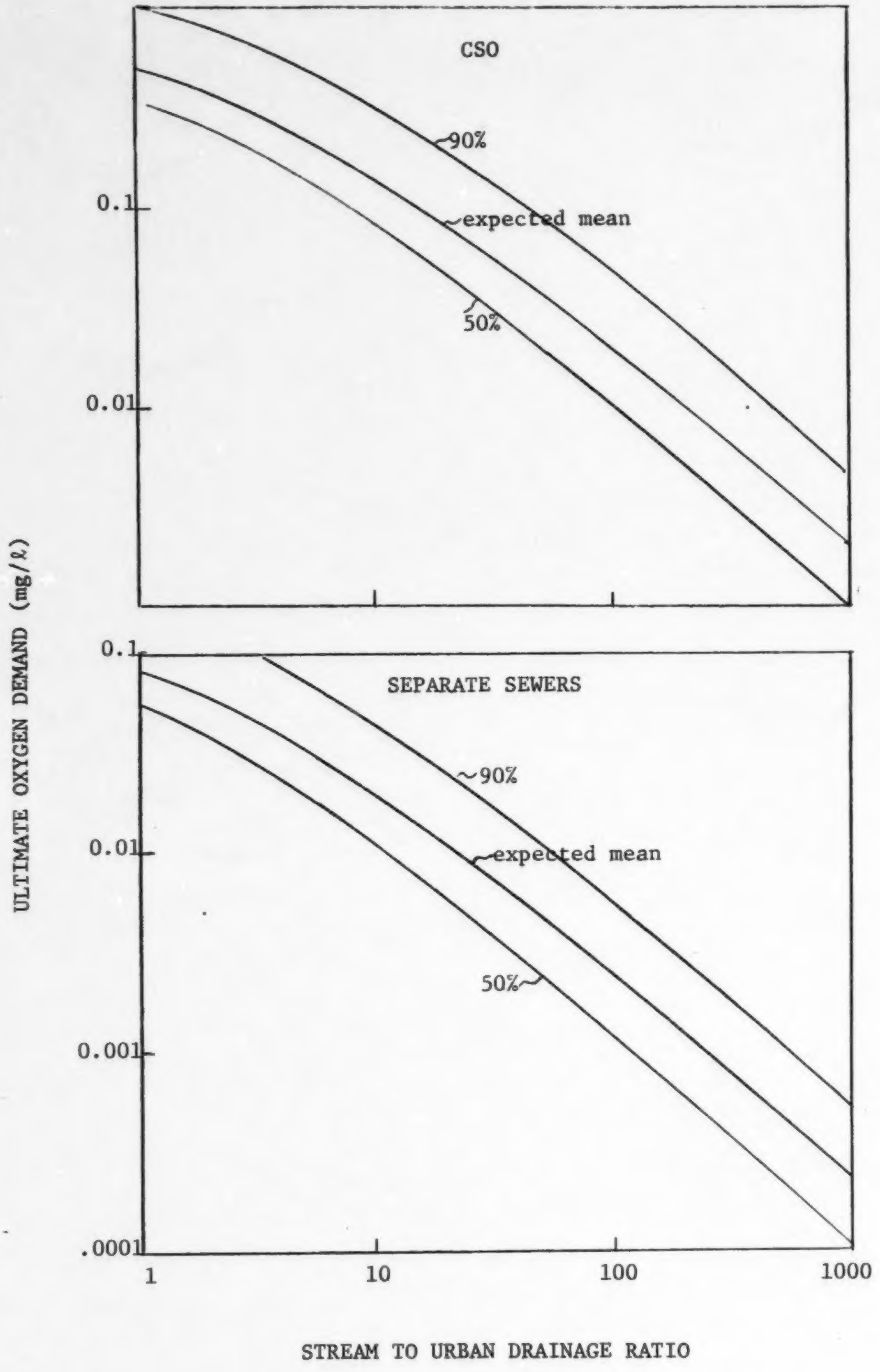


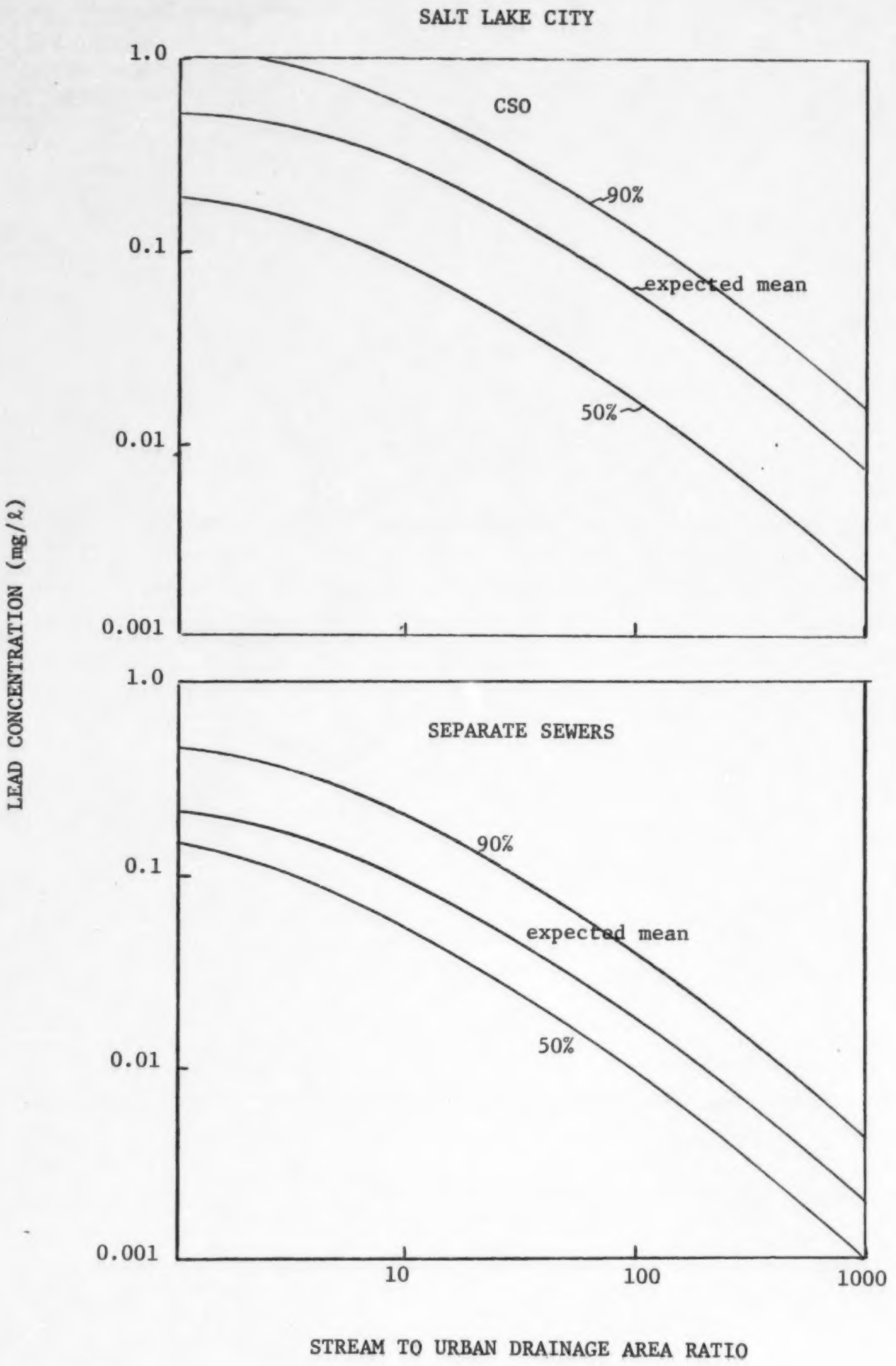
STREAM TO URBAN DRAINAGE AREA RATIO

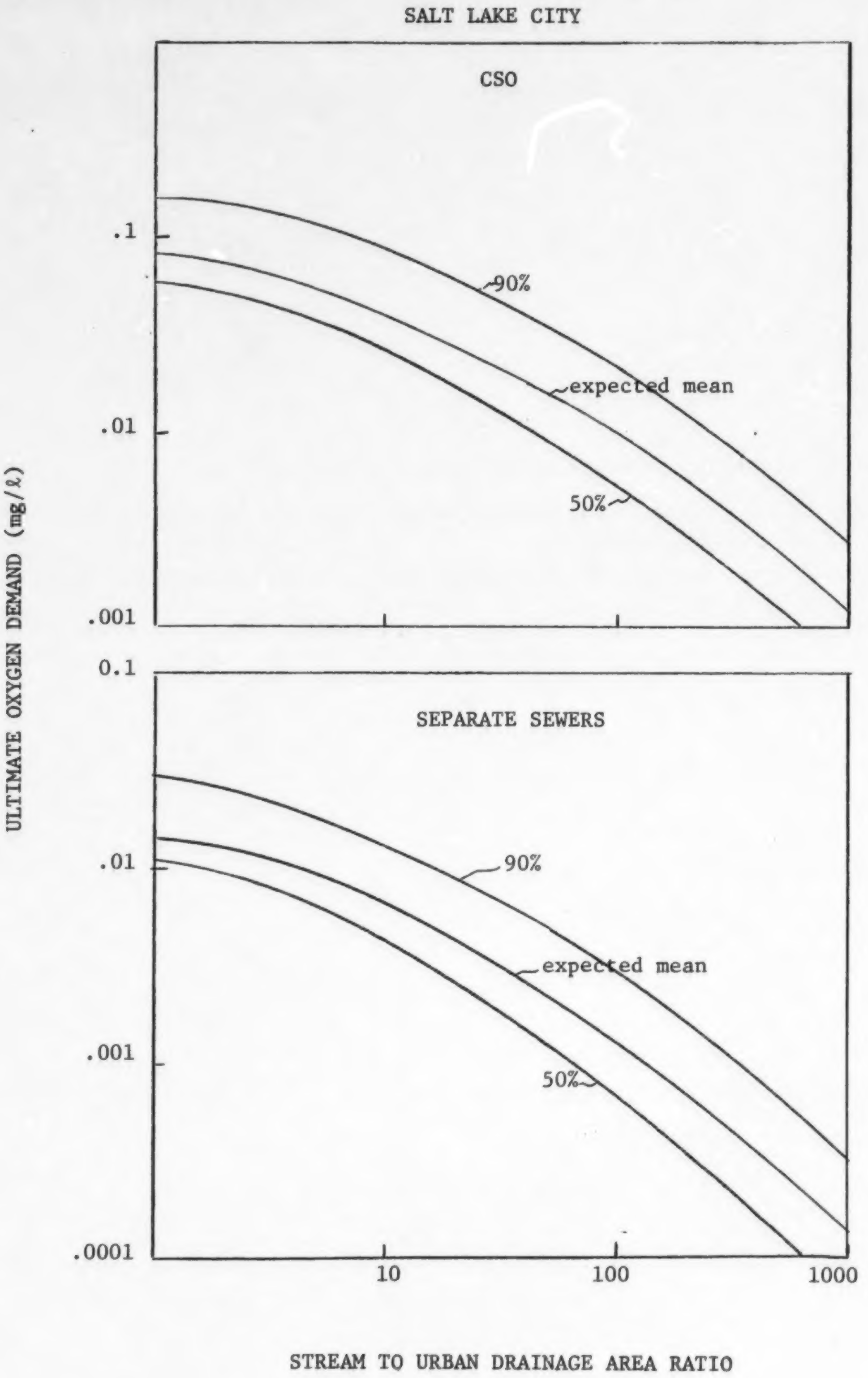
PORTLAND-3 (area c)



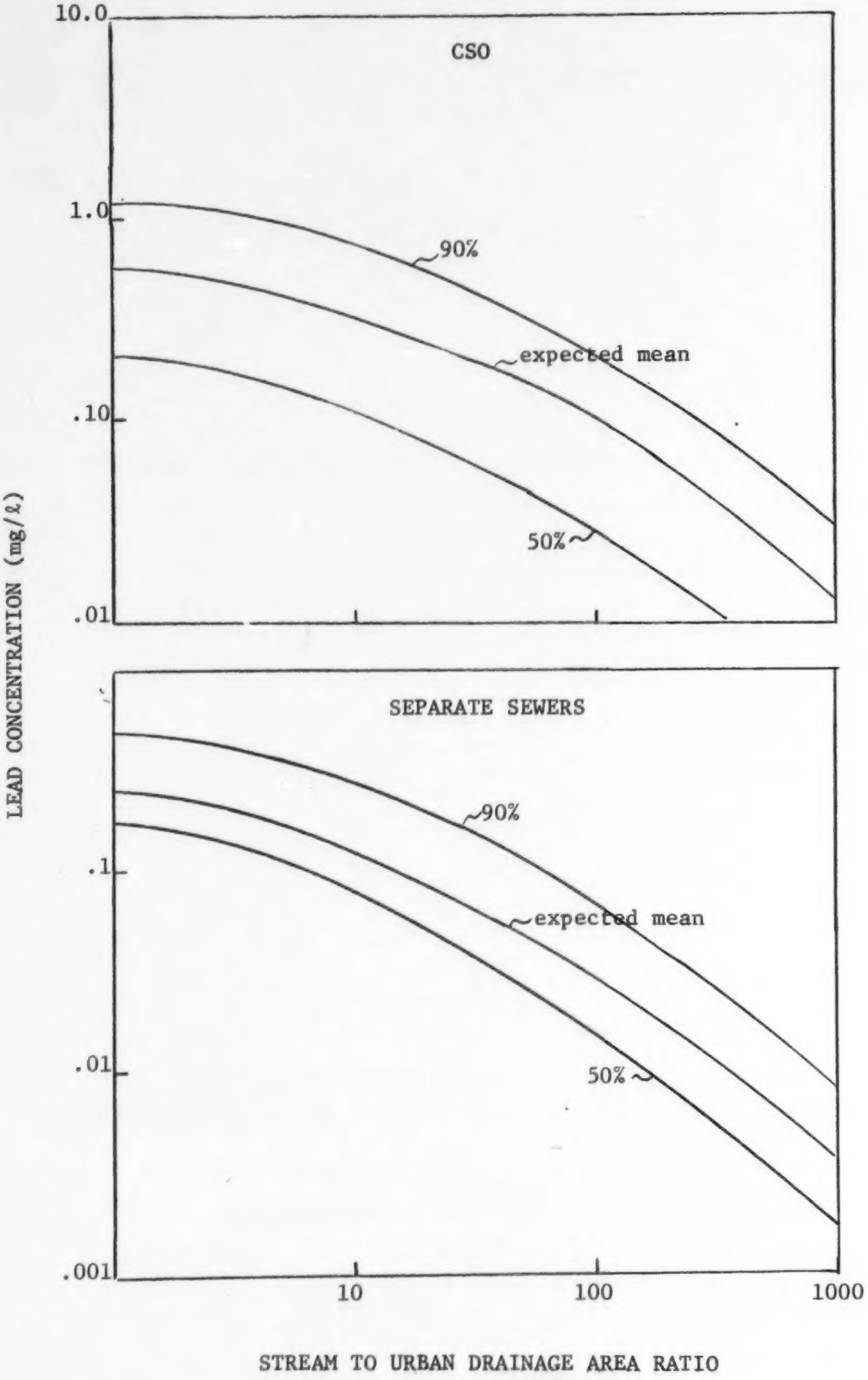
PORTLAND-3 (area c)




SALT LAKE CITY
CSO
90%
expected mean
50%
SEPARATE SEWERS
90%
expected mean
50%
LEAD CONCENTRATION (mg/ℓ)
1.0
0.1
0.01
0.001
1.0
0.1
0.01
0.001
10
100
1000
STREAM TO URBAN DRAINAGE AREA RATIO

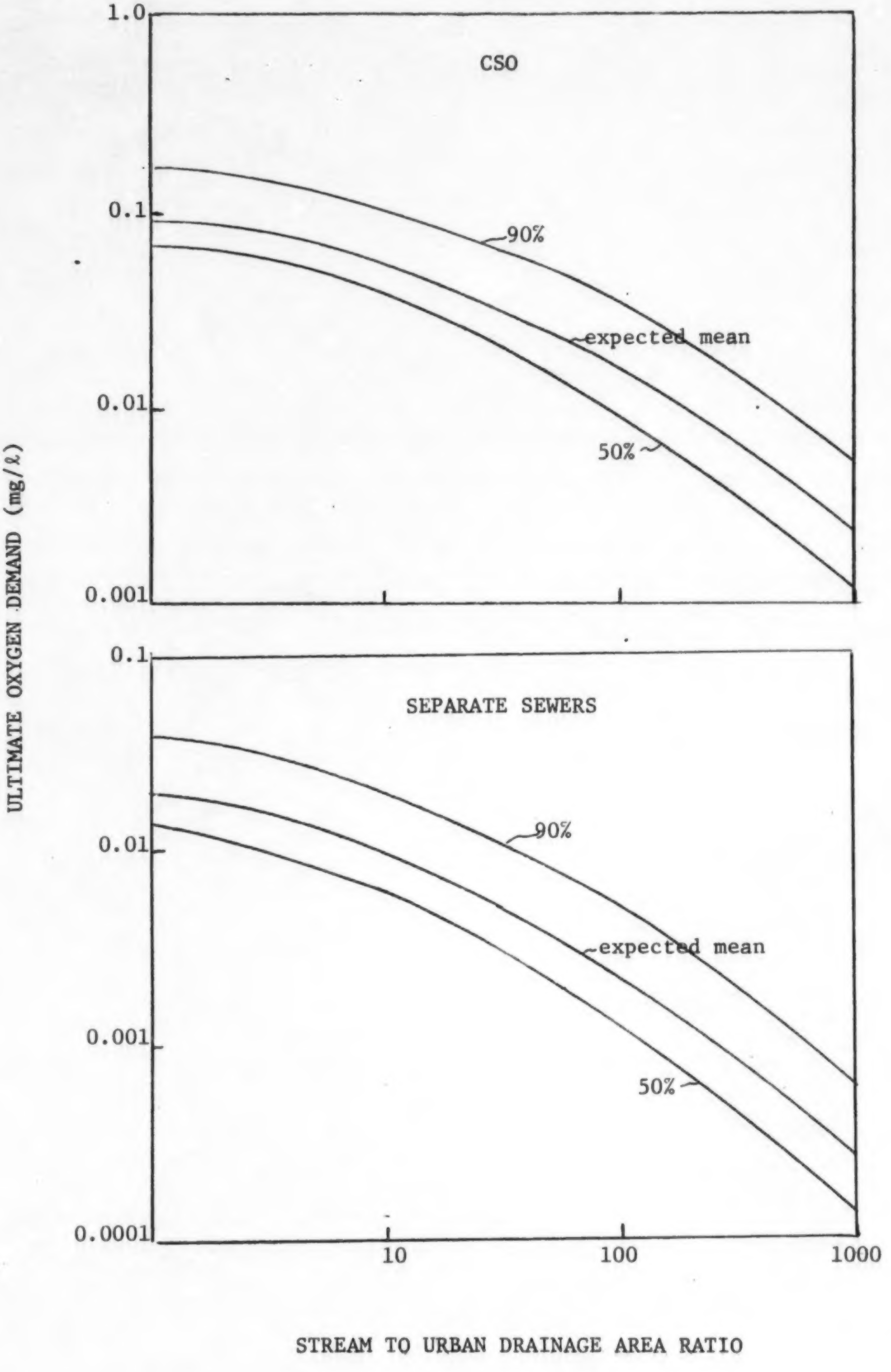

SALT LAKE CITY
CSO
90%
expected mean
50%
SEPARATE SEWERS
90%
expected mean
50%
ULTIMATE OXYGEN DEMAND (mg/ℓ)
.1
.01
.001
0.1
.01
.001
.0001
10
100
1000
STREAM TO URBAN DRAINAGE AREA RATIO

TAMPA



CSO
10.0
1.0
.10
.01
90%
expected mean
50%
SEPARATE SEWERS
.1
.01
.001
90%
expected mean
50%
10
100
1000
LEAD CONCENTRATION (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO

TAMPA



1.0
0.1
0.01
0.001
CSO
90%
expected mean
50%
0.1
0.01
0.001
0.0001
SEPARATE SEWERS
90%
expected mean
50%
10
100
1000
ULTIMATE OXYGEN DEMAND (mg/ℓ)
STREAM TO URBAN DRAINAGE AREA RATIO